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Habitat Value of Constructed and Natural Wetlands Used to Treat Urban Runoff: A Literature Review

A Report Prepared for the California State Coastal Conservancy

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EXECUTIVE SUMMARY

With the rising urbanization of the coastal watersheds of southern California, wetlands and riparian areas have been rapidly disappearing from the landscape, and those that remain are often highly degraded. Concurrent with the loss of wetland habitat, increased runoff from urbanized watersheds and discharges of nonpoint source pollution have created a demand for effective, low-cost solutions to improve surface water quality and attenuate storm flows. As a result, there is increasing pressure to restore, enhance, and create natural or constructed wetlands with multiple objectives (i.e., habitat support, treatment of nonpoint source pollution, flood attenuation, and recreation). Those who manage wetlands for multiple objectives often claim that all these benefits accrue in treatment wetlands. Many of these same managers and other authors have asserted that use of wetlands for treatment purposes may pose a threat to wildlife that should be accounted for in design and siting. The extent to which multiple benefits are actually provided, the actual risk to wildlife, and the trade-offs between these different objectives is often not clear.

To better understand the risks, considerations, and trade-offs associated with using wetlands to treat urban runoff, we reviewed the existing literature on the subject and identified the state of knowledge and research gaps on the issue. The need for this work was identified by the California Coastal Conservancy and the Southern California Wetland Recovery Project (WRP), which have received several requests to fund urban wetland restoration projects whose stated objectives include improving the water quality of urban runoff as well as restoring riparian and wetland habitat. The intended use of this review is to provide a baseline of information from which the WRP partner agencies may begin to address key research, management, and policy questions pertaining to the ability of wetlands to fulfill the dual objectives of treating urban runoff and providing wildlife habitat, given the uncertainties.

To better understand the complex issues associated with the habitat quality or hazards associated with natural or constructed wetlands subject to urban runoff, literature was reviewed to address the following questions:

1) Do natural or constructed wetlands used for treatment of urban runoff become environmental hazards for wildlife?
2) How does directing urban runoff or stormwater through or a natural or constructed wetland affect the habitat of the wetland?
   a) What is the potential that changes to a wetland's physical or chemical structure will affect the ecological health of the wetland, in terms of its ability to support a diverse and healthy assemblage of typical wetland flora and fauna?
   b) How do potential changes in wetland physical and chemical structure impact the biological communities dependent on these wetlands, and are these impacts severe enough to be of concern?
   c) To what extent does the intentional manipulation of hydrologic regime, site morphology, and wetland biota to maximize water treatment capacity detract from the habitat value of the wetland?
3) How do considerations of the habitat effects of treatment wetlands differ by wetland class (e.g., how are the issues different for riverine vs. palustrine or lacustrine wetlands)?

Review of over 200 reports, articles, books, and other references allowed us to draw the following general conclusions:

- *Adequate research does not exist to address the three central questions given above.* The existing research literature is generally tangential to the central topics of interest. Studies generally address either the efficacy of treatment wetlands in removing contaminants or the effects of urbanization on wetland attributes. However, there is little to no literature that directly addresses the questions posed above.

- *Documented examples exist of potential incompatibility between water quality and habitat goals in wetlands (e.g., Kesterson Marsh). Overall, the literature reviewed suggests concerns of risk to wildlife are valid, but because these studies were not specifically designed to address this question, the degree of risk is unknown.* Research on the effects of urbanization on wetlands has found that urban runoff can affect the biotic communities that utilize these wetlands by altering the hydrology, biogeochemistry, diversity of species and habitat types, community structure, and trophic level dynamics. Currently in southern California, many of the remaining freshwater wetlands are sustained by urban runoff. It is not clear however, given the extreme loss of aquatic habitat in southern California, that the risks posed by such a use are outweighed by the additional benefit provided by the restoration or creation of aquatic habitat (albeit for water quality purposes).

- *There is a general lack of literature on the efficacy and effects of using wetlands to treat urban runoff in semi-arid/climates such as southern California.* Most of the literature that addresses efficacy of treatment or effects of urbanization on wetlands has been conducted in humid, temperate climates. This literature would be helpful in addressing the risk posed to wildlife from treatment wetlands, but is less applicable to local conditions given southern California's semi-arid climate.

There is extensive literature that discusses how the physical and biological structure of a wetland affects either water quality or habitat function. However, knowledge of how to (or whether it is advisable to) optimize a wetland to meet both these objectives is poor. To begin addressing this question, we recommend three general areas for additional research:

- Investigation of how increased inputs of urban runoff to wetlands affects the physical and chemical structure of the wetland;
- Investigation of how increased urban runoff and associated loadings affects wetland plants, animals, and community structure; and/or
- Investigation of treatment wetland siting, design, and management strategies that mitigate impact to wildlife habitat, and the trade-offs or synergy with water quality objectives (a preliminary list of strategies from the literature is provided in Chapter 5).

To address these questions, a risk characterization needs to be conducted on several treatment wetlands and compared to natural wetlands in comparable landscape settings. A list of specific research questions that should be addressed in this risk characterization is provided in Chapter 6.

Siting Design and Management of Treatment Wetlands: Summary of Chapter 5

Landscape Planning Considerations

Siting of treatment wetlands should be part of a watershed-wide planning effort that includes a comprehensive water quality management strategy

- Maximize water storage and infiltration opportunities outside of existing wetlands to minimize runoff.
- Consider decentralized treatment wetlands throughout the watershed over one large treatment wetland.
- Research historic extent of natural wetlands within watershed and locate treatment systems where there is minimal opportunity to restore historic or natural wetlands.
- The installation of treatment wetlands in a watershed should be coupled with a program to preserve open space and mature forest cover and restore riparian buffers around streams and wetlands.
- In wetlands and streams whose hydrology has been disturbed, consider managing stormwater and low flow runoff to match, as close as possible, the predevelopment hydroperiod and hydrodynamic.
- Since evidence suggests that urban runoff may affect natural wetlands, consider strategies that improve urban runoff quality before it enters natural wetlands or aquatic habitats.
- If habitat benefits are desired, then assess whether the water to be treated or sediments are of adequate quality to support wildlife.

Design Considerations

A well-conceived design for a constructed wetland is critical not only to mitigate any potential impacts to wildlife but also to assure a high level of treatment performance.

- Hydraulic and Contaminant Loading Rates: Given lack of guidelines for southern California, size treatment wetlands to allow for conservative hydraulic and contaminant loading rates.
- Importance of Source Control and Pretreatment: Literature suggests that wetlands should not be the first interceptor of urban runoff, particularly in industrial or highly urbanized sites. Specific recommendations include:
  - Control source of pollutants where possible
  - Depending on land use, provide pretreatment including: 1) oil and grit interceptors in highly industrial sites, highways, etc. , 2) sand filters or forebays and floating berms to trap trash and large debris prior to or at beginning of wetland such that the remainder of the wetland is used for polishing and/or wildlife enhancement, and 3) incorporate a variety of treatment strategies in series to maximize removal efficiencies and minimize exposure to wildlife.
- Maximize Native Habitat and Plant Diversity: Because the plant palette is one of the major controls on habitat quality in a treatment wetland, the following recommendations be considered in the design:
  - Where appropriate, design constructed treatment wetland to provide habitat with a diversity of native species comparable to similar wetlands in the region.
  - Create gentle slope to allow for good plant establishment and diversity.
  - Design for moderate water level fluctuations.
  - Maximize vegetative species diversity, where appropriate. Research is needed to determine which native upland and wetland plant species are compatible for treatment wetland use.

Maintenance

Long-term, regular maintenance of treatment wetlands is critical to sustain treatment capacity and optimize the habitat value provided and should be required indefinitely. All maintenance work must be scheduled to avoid critical breeding and nesting periods for wetlands species.

Importance of Monitoring and Adaptive management

US EPA strongly recommends long-term monitoring of treatment wetlands to ensure that the system is functioning properly and not becoming an attractive nuisance to wildlife. See Chapter 5 for more details.

Research Recommendations: Summary from Chapter 6

Below is an initial list of research needs required to assess the potential hazards associated with treating wet and dry season urban runoff in wetlands of arid/semi-arid climates. A risk assessment paradigm is useful to examine these issues and consists of two components: (1) an exposure assessment quantifies the potential level of damage to the habitat and (2) an ecological effects assessment evaluates the effect of the damage or contamination to a particular species or community. Given that different agencies or organizations may have different priorities, no attempt was made to prioritize the list of research needs. Rather, prioritization of a research agenda should be done via consensus involving all appropriate entities.

Exposure Assessment

1. What are wet and dry season pollutant concentrations and loading rates from specific land uses?
2. What is effluent quality of treatment wetlands (constructed and natural) and riparian areas in semi-arid/arid climates under a range of contaminant loading rates and over time?
3. What are the natural background concentrations of contaminants in the surface waters of wetlands and riparian areas in southern California? How do they vary spatially (both by position within the watershed and between watersheds) and temporally over seasonal or climatic cycles?
4. How does diversion of wet or dry season flow into wetlands and riparian areas determine 1) physiochemical characteristics (pH, conductivity, alkalinity), 2) sediment deposition rates, texture and grain size distribution, and organic matter content, 3) hydrologic regime including seasonal and annual water budgets, 4) storage, transport, and seasonal/annual budgets of contaminants?
5. What is the maximum allowable change in peak flow rate and duration, water level, and other hydrologic variables that can result without deleterious effects to habitat values (by wetland class)?

Ecological Effects Assessment

1. What are the most appropriate assessment endpoints to evaluate risk (e.g., plant chlorosis, decreases of microbial communities, changes in species composition, reproductive failure)?
2. How does biotic community composition/trophic structure compare between treatment vs. natural wetlands?
3. What is the sensitivity of key plant or animal species to contaminant loading and accumulation?
4. What are the rates of bioaccumulation of contaminants at various trophic levels?
5. Is toxicity observed in macroinvertebrates, fish, amphibians, or birds that live, forage, or breed in wetlands exposed to urban runoff? Are specific life stages or species more sensitive than others? What are the most appropriate biological indicators for monitoring toxicity?
6. What is the habitat value of treatment wetlands on a landscape scale?

Mitigating Risks to Wildlife and Maximizing Habitat Benefits

Research on the effect of specific design or maintenance practices on wildlife will improve our ability to optimize wetland management for both wildlife and water quality goals.

1. What are the appropriate design criteria for wetlands treating wet weather and dry weather urban runoff if providing habitat is a primary or secondary objective? What are the appropriate design criteria if the objective is to construct a treatment wetland and not attract sensitive species?
2. What other treatment wetland design attributes contribute to improving habitat in wetlands?
3. What is the effect of pretreatment on reducing contaminant loading to the main wetland or riparian area? What is the extra burden in cost and labor that such pretreatment strategies impose?
4. To what extent does routine maintenance in forebays and main wetland cause a disturbance or diminish habitat value? What are recommended ways to minimize this disturbance?
5. What plant species native to southern California are most suitable for cultivation in treatment wetlands (i.e., the most flood and contaminant tolerant)? What are the recommended native plant palettes with respect to different habitat types within a treatment wetland?
6. What source control/treatment BMPs are appropriate as part of an overall water quality program?

CHAPTER 1.
INTRODUCTION AND PURPOSE OF LITERATURE REVIEW

A. Background and Purpose of Literature Review

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 (CDPR 1988). Impacts have been particularly severe for coastal salt marshes (CCC 1989; CDFG 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, low-cost 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)

The purpose of this literature review to assemble current information and identify data and research gaps on the habitat benefits and risks associated with the use of natural, constructed wetlands, or restored wetlands for improving the water quality of urban runoff. Specifically, we are interested in a) the potential habitat effects of diverting or increasing urban runoff to natural systems (e.g., changes in water quality, hydrology, or physical structure) and b) the potential value and risks to wildlife of habitat that is provided by treatment wetlands.

The need for this work was identified by the California State Coastal Conservancy and the Southern California Wetland Recovery Project (WRP)[1], which have received several requests to fund urban wetland restoration projects whose stated objectives include improving the water quality of urban runoff as well as restoring riparian and wetland

[1] The Southern California Wetland Recovery Project (WRP) is a partnership of 17 state and federal agencies along with local non-profits working to develop and implement a regional plan for wetland acquisition, restoration, and enhancement in southern California. A mission statement and list of WRP partners can be found on the California Coastal Conservancy website (http://www.coastalconservancy.ca.gov/scwrp/index.html).

habitat. In addition, several regulatory agencies have identified the need for additional information on the implications of using urban runoff to support wetlands created or restored as compensatory mitigation and/or the advisability of placing treatment wetlands adjacent to natural habitat areas. The intended use of this review is to provide a baseline of information from which the WRP partner agencies may begin to address key research, management, and policy questions pertaining to the ability of wetlands to fulfill the dual objectives of treating urban runoff and providing wildlife habitat, given the uncertainties.

B. Key Definitions and Concepts

In undertaking this literature review, it is important to define key terms and concepts such as "wetlands" and "riparian areas" or "constructed" or "natural" wetlands that are used throughout this document. The definition of wetland and riparian areas, as well as the distinction between "natural" and "constructed" wetlands are given below.

Definition of Wetlands and Riparian Areas

Wetlands are zones that lie on a continuum between terrestrial and aquatic environments, and demarcation of the boundaries often is not clear-cut. For the purpose of this document, we utilize the US Fish and Wildlife Service definition of wetlands:

> *"Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. For the purposes of this classification wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is predominantly undrained hydric soil; and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year"* (Cowardin et al. 1979).

The wetland ecosystems in southern California coastal watersheds include four types or systems: estuarine, riverine, lacustrine and palustrine wetlands (Cowardin et al. 1979). They are defined as follows:

- *Estuarine* - Deepwater tidal habitats and adjacent tidal wetlands that are usually semi-enclosed by land but have open, partly obstructed, or sporadic access to the ocean, with ocean water at least occasionally diluted by freshwater runoff from the land. The upstream and landward limit is where ocean-de rived salts measure less than .5 parts per thousand during the period of average annual low flow.
- *Riverine* - All wetlands and deepwater habitats contained within a channel except those wetlands (1) dominated by trees, shrubs, persistent emergents, emergent mosses, or lichens, and (2) which have habitats with ocean-derived salinities in excess of .5 parts per thousand.
- *Lacustrine* - Wetlands and deepwater habitats (1) situated in a topographic depression or dammed river channel; (2) lacking trees, shrubs, persistent emergents, emergent mosses, or lichens with greater than 30% areal coverage; and (3) whose total area exceeds 8 hectares (20 acres); or area less than 8 hectares if the boundary is active

wave-formed or bedrock or if water depth in the deepest part of the basin exceeds 2 m (6.6 ft) at low water. Ocean-derived salinities are always less than .5 parts per thousand.

- *Palustrine* - All nontidal wetlands dominated by trees, shrubs, persistent emergents, emergent mosses, or lichens, and all such tidal wetlands where ocean-derived salinities are below .5 ppt. This category also includes wetlands lacking such vegetation but with all of the following characteristics: (1) area less than 8 ha; (2) lacking an active wave-formed or bedrock boundary; (3) water depth in the deepest part of the basin less than 2 m (6.6 ft) at low water; and (4) ocean-derived salinities less than .5 parts per thousand.

Riparian areas may be referred to as systems, zones, corridors, or habitats and have been defined in a variety of ways, which can be somewhat confusing. The US EPA defines a riparian ecosystem as:

> *"…a vegetated ecosystem along a water body through which energy, materials, and water pass. Riparian areas characteristically have a high water table and are subject to periodic flooding and influence from the adjacent water body. These systems encompass wetlands, uplands, or some combination of these two landforms. They will not have in all cases the characteristics necessary for them to be also classified as wetlands*" (USEPA 2001).

The National Research Council reviewed seven definitions of "riparian" currently used by different Federal Agencies and developed the following definition:

> *" . . . transitional areas between terrestrial and aquatic ecosystems that are distinguished by gradients in biophysical conditions, ecological processes and biota. They are areas through which surface and subsurface hydrology connect waterbodies with their adjacent uplands. They include those portions of terrestrial ecosystems that significantly influence exchanges of energy and matter with aquatic ecosystems. Riparian areas are adjacent to perennial, intermittent, and ephemeral streams, lakes, and estuarine-marine shorelines"* (NRC 2002).

The key aspect of these definitions is that riparian areas are transitional between upland and aquatic ecosystems and must be considered in the context of the adjacent habitat types. These areas may be perennial, intermittent, or ephemeral, can support aquatic or mesophytic vegetation (trees, scrub and herbaceous cover), and may have predominantly nonhydric soils. Riparian areas are not just unique to the upland transition zones of riverine wetlands (in linear corridors), but can also be found adjacent to palustrine, lacustrine and estuarine wetlands. The location of riparian areas between upland catchments and their receiving waters often sets up potential conflicts between their intended uses as habitat and treatment areas for urban runoff. For the purposes of this literature review, our use of the term "wetlands" will include "riparian areas".

Distinction between "Natural" and "Constructed" Wetlands

In defining the scope of this literature review, it is important to understand the distinction between "natural" and "constructed" wetlands, and how this potentially affects the habitat value they provide. According to Kadlec and Knight (1996), a natural wetland is one which "occurs without the aid of humans", while a constructed wetland is one which is "purposely constructed by humans in a non-wetland environment." Constructed wetlands designed to optimize water quality benefits may provide wildlife benefits. Conversely, natural wetlands, often restored or enhanced to optimize wildlife habitat, may also provide water quality benefits.

While on paper this distinction seems perfectly clear, in reality the range of wetlands occurring in southern California cannot be easily lumped into these two categories; rather it is more accurate to characterize them as falling along a continuum from "natural" to "constructed." For example, in a highly urbanized environment, restoration at a site in which natural wetlands may have once occurred may be greatly constrained by human infrastructure. As a result, restoration of a wetland to that site may only occur through engineered solutions (e.g., excavation of upland soils, engineered physical structures, diversion of stormwater or publicly-owned treatment works (POTW) effluent to provide a stable water source). Similarly, the physical and biological structure of "natural" streams may be altered by changes in runoff or sediment generation associated with the altered hydrologic regime of an urban environment. These wetlands, though "natural" can often take on attributes of "constructed" wetlands, while constructed wetlands, particularly when located in a less developed setting and passively managed, often take on the attributes of a "natural" wetland. Engineers, scientists, and managers who create, restore, or enhance wetlands for multiple objectives face a series of design decisions that affect the balance of benefits achieved for each objective.

The capacity of a wetland to support wildlife is likely dictated more by the structure and condition of the habitat than by whether the wetland is "natural" and/or "constructed." Furthermore, owners or managers of wetlands constructed for either habitat or water treatment purposes are responsible for the water quality in the wetland and are potentially liable for negative impacts to threatened and endangered species and migratory birds sustained through their use of this habitat (Lemly and Ohlendorf 2002). Therefore, it is important to characterize the benefits <u>and potential risks</u> to wildlife of both natural and constructed wetlands receiving urban runoff.

C. Organization of Review and Criteria for Selecting Literature

Organization of Review

This literature review is organized in six chapters. **Chapter 1 g**ives an introduction to the overall issue, gives key definition, and lays out the organization of review and criteria for selecting literature.

The purpose of **Chapter 2** is to provide general background information on the factors that affect a wetland's treatment capacity (whether natural or constructed), to review some of those factors that can be engineered to maximize that capacity, and to address some specific design considerations for wetlands used to treat stormwater and dry-season urban runoff.

Chapter 3 addresses the factors affecting the habitat value of wetlands used to treat urban runoff with respect to the physical structure of habitat (hydrology, site morphology, sediment transport, surface water, and sediment chemistry).

Chapter 4 addresses the factors affecting the habitat value of wetlands used to treat urban runoff with respect to effects on biological communities (ecotoxicological effects, species and habitat diversity, etc.).

Chapter 5 provides a summary of suggestions from the literature on the measures that can be taken to minimize adverse effects to natural and constructed wetlands from a design and maintenance perspective.

Chapter 6 provides a summary of the findings of the literature review and highlights the gaps in research and data needed to address major technical issues and uncertainties in utilizing natural and constructed wetlands in southern California for the dual purpose of providing habitat support and water quality improvement.

Criteria in Selecting Literature

Articles selected as a basis for this literature review generally fell within the following criteria:

- Peer reviewed journal articles
- Federal and state government documents
- Emphasis on semi-arid and arid climates (when possible)
- Literature on natural wetland physical, chemical and biological processes
- Literature on surface flow constructed treatment wetlands[2]
- Emphasis on urban runoff and nonpoint impacts to natural and constructed wetlands.

[2] Constructed wetlands are generally categorized into two types: surface flow (SF) and subsurface flow (SSF). With SF constructed wetlands, water surface is exposed to the atmosphere and the plants are rooted at the bottom of the wetland in some type of substrate. With SSF constructed wetlands, plants are not submerged in water but water typically flows horizontally through a gravel substrate allow a large surface area for microbial activity and growth. Natural wetlands and most constructed wetlands used to date for treatment of urban runoff are surface flow, so this review generally excludes literature on SSF constructed wetlands.

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CHAPTER 2. REVIEW OF BASIC CONCEPTS: FACTORS AFFECTING WATER QUALITY TREATMENT CAPACITY OF WETLANDS

Although natural wetlands have been used as wastewater discharge sites in some cases for hundreds of years, recognition of the water quality treatment capacity of wetlands has emerged only in the last 40 years as monitoring was initiated at existing discharge sites (Kadlec and Knight 1996). Over the past four decades, there has been a tremendous growth in the research of the factors that affect the water quality treatment ability of wetlands, and the engineering of constructed wetlands to maximize this ability. As a result, there is a wealth of literature and data available that summarizes current knowledge on the fundamental processes, efficiency of treatment under a variety of conditions, and design concepts of constructed wetlands (Hammer 1989; Hammer and Kadlec 1983; Kadlec and Knight 1996; Knight et al. 1999; Reed 1990), particularly for municipal and industrial wastewater discharges. While the body of literature studying the efficacy of wetlands to treat urban runoff is growing, much less information exists on design criteria and efficacy than with those treating municipal wastewater.

The purpose of this section is to provide an overview of the factors affecting the treatment capacity of wetlands, describe some of the strategies used to optimize treatment capacity, and also describe some of the specific considerations necessary in using wetlands to treat urban runoff. The purpose of this chapter is not to provide a comprehensive overview, but rather to present a brief summary of the concepts and provide key references where the reader may go for more information.

A. <u>Factors Affecting Fundamental Biogeochemical Processes in Wetlands</u>

The cycling of all compounds (including natural and anthropogenic sources of contaminants) in natural and constructed wetlands is controlled by several fundamental mechanical, chemical and biological processes (Figure 1). These mechanisms include: 1) the flocculation, settling and filtration of suspended matter and floatables, 2) volatilization, 3)) adsorption and desorption of compounds from particles, 4) chemical dissolution and precipitation between water-borne solutes and solids, 5) chemical transformations, and 6) biological uptake, transformation and release (Mitsch and Gosselink 1993; Tschobanoglous 1993). Together, these processes form a complex set of interactions responsible for the water quality treatment functions of wetlands. The relative importance of these processes, and thus the degree to which these functions enhance water quality, is dependent on the wetland hydrologic regime, the geology and chemistry of sediments, the composition of the microbial community, the flora and fauna that inhabit the wetlands, and the geomorphology and position of the wetland in the landscape.

Hydrologic Regime

The hydrology of a wetland is one of the most defining characteristics affecting physical structure, sediment characteristics and biota, and therefore is one of the most important factors affecting a wetland's treatment capacity. The relative magnitude and timing of flow through a wetland via surface water inflow and outflow, groundwater inputs and outputs, rainfall, and evapotranspiration controls the volume, depth, duration and frequency of flooding, and residence time of water in a wetland (Kadlec 1989; Watson and Burnett 1995). These factors in turn control the treatment capacity by affecting the duration of interaction between the water column and the sediments/biota, the proximity of contaminants to the sites of biogeochemical activity where some forms of treatment occur, and the oxygen levels in the sediment and water columns (Craft 2001; Kadlec and Knight 1996).

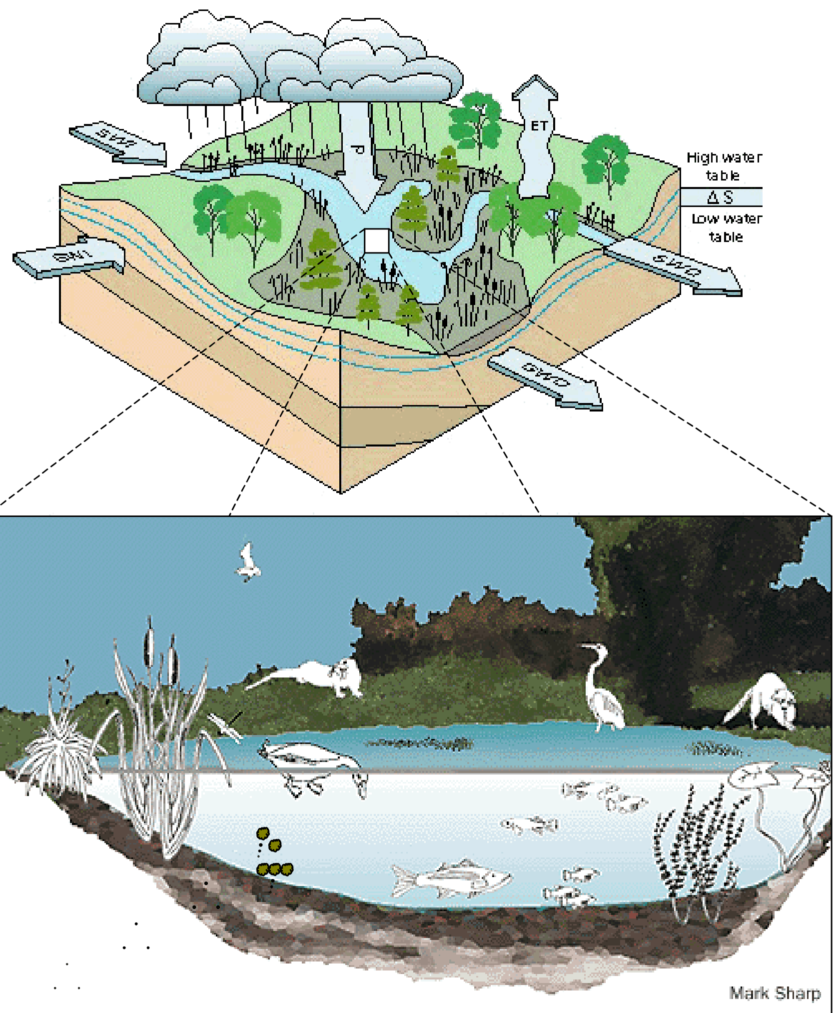

FIGURE 1. Biogeochemical processes that improve water quality in wetlands shown in the above schematic include: (A) flocculation and settling of suspended matter from the surface water; (B) adsorption and desorption of compounds from particles in suspended matter and sediment; (C) chemical dissolution, precipitation, and transformations between contaminants in surface water and sediments; and (D) biological uptake, transformation and release in vascular plants, algae, microbes, and large fauna.

The duration of the interaction between water column and sediments/biota is controlled primarily by hydraulic residence time or detention time, which is defined as the volume of water storage divided by the rate of water flow into the system. Longer residence time increases the opportunities for contact with biogeochemically active sites, so greater removal of waterborne contaminants can occur (Kadlec 1989). When coupled with low velocity flows, long residence time also enhances sedimentation and flocculation of contaminants associated with suspended matter. It also increases accumulation of organic carbon, which fuels many microbially mediated transformations. Consequently, treatment capacity is higher in wetlands where water flows overland through vegetation versus in channels (Delaune and Gambrell 1996; Stumpf 1983; Wolanski 1995).

Contact time with biogeochemically active sites is also affected by wetland storage volume, depth, and duration and frequency of flooding. In general, permanently flooded wetlands have more stable hydrologic and biogeochemical regimes, leading to more efficient pollutant removal over the long term (Kadlec and Knight 1996). Large fluctuations in hydroperiod or flow rate (as is typically seen in natural wetlands) may lead to temporal and spatial variability in volume and water depth. This may result in periods of seasonal and interannual patterns of drying and associated changes in plant species composition (Thibodeau and Nickerson 1985). While this variability is an inherent condition in natural wetlands, it may result in less predictable treatment efficiency, depending on the pollutant.

Geology and Chemistry of Wetland Sediments

Sediments provide the physical platform for many of the biotic and abiotic interactions within a wetland. The physical and chemical nature of the sediments found in a wetland (and their effect on treatment capacity) are the product of the geologic parent materials (including bedrock and sedimentary deposits) which are modified over time by physical chemical, and biological processes within the wetland and its contributing watershed (e.g., weathering and dissolution, Likens et al. 1977). Characteristics of wetland sediment that influence the overall treatment capacity of a wetland include the extent and distribution of aerobic and anaerobic zones, the quality and quantity of mineral and organic matter, and the texture (bulk, density, porosity) and particle size distribution (Faulkner and Richardson 1989).

Wetland sediments are generally distinguished from upland soils by lower levels of oxygen and redox potential (a parameter measuring the potential for a series of oxidation and reduction reactions to occur in the sediments) caused by intermittent or permanent flooding (Gambrell and Patrick 1978). As the duration of flooding increases, anaerobic conditions begin to predominate, slowing the decomposition of organic matter. As a result, wetland sediments generally have higher organic content than upland soils. Near the sediment-water interface, interaction with oxygenated surface waters or the presence of benthic algae or macrophytes creates a layer of aerobic sediment (Armstrong 1964; Moorhead and Reddy 1988). Below this zone, oxygen content decreases, and anaerobic conditions are prevalent. The vertical gradient of these aerobic and anaerobic zones in the sediment control a series of chemically- and biologically-mediated oxidation and reduction reactions that are critical in the sequestration and transformations of nitrogen, phosphorus, heavy and trace metals (Gambrell and Patrick 1978).

The particle size distribution of wetland sediments ranges in size from fine-grained silts and clays to coarse sands. The higher surface areas associated with fine-grained silt and clays generally have a greater number of sorption sites per unit surface area, higher organic matter content and higher cation exchange capacity. These characteristics give fine-grained particles a higher capacity to bind contaminants such as heavy metals, orthophosphates, and some pesticides (Gambrell 1994; Gambrell et al. 1991; Gambrell and Patrick 1978). Although the concentration of organic matter in wetland soils it is generally higher than what is observed in upland soils, the level can vary greatly among different wetland types. For example, ephemeral streams in arid environments typically have relatively low soil organic matter; consequently the cation exchange capacity (the number of negative charges on the surface of soil that can attract positively charged cations) and ability to sequester charged compounds (e.g., heavy metals) is lower than in palustrine wetlands.

Composition of the Microbial Community

Many of the chemical transformations (primarily of organic compounds) that occur in wetlands are mediated by microbial organisms, primarily bacteria and fungi. Microbial transformations occur in surface water, on suspended solids, and in wetland sediments (Alongi 1995; Fenchel et al. 1998; Robertson 1988; Ward and Wetzel 1984). Transformations may occur via direct sorption and transformation of dissolved constituents. They can also occur via a symbiotic relationship with wetland plants and animals whereby they assist in the capture and assimilation of dissolved compounds, making them available to their host. Bacteria are important agents of both aerobic and anaerobic decomposition, whereas fungi are primarily aerobes or facultative anaerobes. In terrestrial environments, microbial transformations are largely limited to aerobic pathways. However, inundated or saturated wetland soils provide both anaerobic and aerobic environments (see previous section). The combination of both these environments greatly enhances wetland's ability to break down organic compounds (Craft 2001).

The physical and chemical condition within a wetland controls a variety of microbial decomposition processes that are critical to the transformation of nutrients and sequestration of trace and heavy metals (Gambrell 1994), including fermentation, denitrification, and methanogenesis. Decomposition rates are highest along the aerobic/anaerobic boundaries, such as the sediment-water interface or across the surface of dissolved organic matter. Therefore, wetlands with more dissolved organic matter, finer-grained substrates, or more complex sediment beds provide more surface area for microbial processes to occur.

Chemical properties, such as temperature, pH, and dissolved oxygen (DO) also affect the efficiency of microbial transformations in wetlands, mainly by influencing the activity of the catalytic enzymes that mediate transformation reactions (Kadlec and Knight 1996). Different microorganisms have varying tolerances, but in general, microbial activity increases with temperature to approximately 35^{o}-40^{o}C, after which microbial activity declines. Most bacteria cannot survive extreme acidity or alkalinity and optimum activity usually occurs between pH 6 and 9. Finally, dissolved oxygen levels in the water and redox potential in sediment affect which microbial communities and metabolic processes dominate (e.g., iron reduction, sulfate reduction), thereby influencing the overall rate and efficiency of transformation processes.

Wetland Flora and Fauna

There is a large body of literature documenting the productivity and species diversity associated with wetlands. This diverse array of wetland biota perform varied and complex roles in the biogeochemical processes of wetlands (Kadlec and Knight 1996). In their cycles of growth and decay, wetland algae and macrophyte, uptake, sequester, transform, and release compounds. When they die, macrophytes provide organic matter to the system which provides substrate for microbial transformations and increases the cation exchange capacity of the sediments (see previous section). Phytoremediation, or the use of algae and plants for the cleanup of contaminants in groundwater, surface waters, upland soils and wetland sediments, takes advantage of efficiency of these organisms to uptake and store contaminants (Conger and Portier 1997; Susarla et al. 1999; Susarla et al. 2002). Invertebrates and vertebrates that utilize wetlands can also play an important role in the biogeochemical processing of pollutants by ingesting organic matter or directly absorbing and transforming contaminants themselves. Bioturbation associated with faunal (i.e. invertebrates, fish and birds) digging, stirring and mixing of sediments often introduces oxygen, influencing the anaerobic/aerobic layering of the sediments; thereby changing the nature of the biogeochemical reactions (Miller 1982; Tudhope and Scoffin 1984).

Landscape Position

The inherent treatment capacity of a wetland is largely dictated by its class (e.g., wetland type) and landscape position (Twilley 1995). Lacustrine and palustrine wetlands are generally more quiescent, and therefore tend to have more stable hydrologic regimes and predictable rates of chemical and biological cycling. Consequently, natural palustrine and lacustrine freshwater marshes and forested wetlands have been used the most for enhancement of surface water quality (Kadlec and Knight 1996). Riverine wetlands and associated riparian areas also have a recognized capacity to improve surface water quality (Mitsch 1995; Nairn and Mitsch 2000). In particular, riparian buffer areas are a widely used best management practice (BMP) to reduce contaminant concentrations in agricultural runoff (Clausen et al. 2000; Fennessy and Cronk 1997). However, riverine wetlands are subject to greater physical energies and variations in hydrology. While removal of some contaminants can be maximized under the aerobic conditions found in riverine wetlands, the variability in hydrology can decrease the predictability of contaminant removal. Seasonal pulsing associated with flood events in riverine wetlands also can remobilize contaminants associated with sediments or vegetation and transport them downstream (Klopatek 1978). Estuarine and marine wetlands, located at the bottom of a watershed, are subject to flooding and episodic sediment transport from their contributing watershed as well as from high tides and storms from the coastal ocean (Mitsch and Gosselink 1993). Gradients in salinity, DO, and chemical composition in estuarine wetlands results in a highly variable environment. Given this variability, the role of estuarine wetlands as a sink, transformer, or source of contaminants is complex and the most debated (Nixon 1980) and their capacity for treatment is highly unpredictable.

B. Optimizing Treatment Capacity of Wetlands

The capacity of a treatment wetland to remove pollutants will vary based on its intrinsic properties, such as its fundamental design, size, hydrologic, sediment, and biotic characteristics (as discussed above), and the watershed conditions, such as climate, loading rate (e.g., land use), and the storm characteristics. Constructed wetlands can vary greatly depending the on design objectives (e.g. whether they are designed to treat dry weather flows only versus dry weather and storm flows). Although natural wetland and riparian areas can be manipulated or managed to improve surface water quality and mitigate the impacts of point and nonpoint source pollution, these benefits should not be maximized at the expense of other wetland functions (e.g., habitat) or in a manner that creates an environmental risk to wildlife (e.g., ecotoxicological). Furthermore, wetlands and riparian areas should not be viewed as a treatment mechanism for unlimited amounts of pollution (USEPA 2001). When the assimilative capacity of a wetland is exceeded, its inherent functions begin to break down and the wetland can become degraded or destroyed. Degraded wetlands (either natural or constructed) lose their capacity to assimilate pollutants and attenuate storm flows (Bedford and Preston 1988; Richardson and Davis 1987) and under certain conditions can actually begin to release previously sequestered material, thereby becoming a source of contamination (Richardson and Davis 1987). Therefore, the goal when designing treatment wetlands should be to optimize the *sustainable* treatment capacity by manipulating hydrology, physical structure, and biota. This is done in consideration of the contaminants of concern in the wastewater, the hydraulic and contaminant loading rates, the treatment and other ancillary goals, as well as other site-specific technical, regulatory and economic restraints (Kadlec and Knight 1996).

There are three principal ways to maximize the treatment capacity of a wetland: 1) increase the duration of interaction between the contaminants and the sites of biogeochemical activity where treatment occurs; 2) increase the surface area of biogeochemically-active sites relative to the contaminant loading rate; and 3) increase, through design or management, the capacity of the system to act as a continuous sink for the contaminants (Kadlec and Knight 1996). How this is done depends on the contaminants targeted for removal, their predominant form (particulate, dissolved organic or inorganic), and the desired mechanism to treat them in the wetland. In all cases, the habitat implications of manipulating a wetland to increase its treatment capacity should be recognized and addressed.

Manipulating Hydrology and Physical Structure of Wetland

Manipulation of the physical structure and the hydrology are the primary tools used by engineers to maximize the efficiency and capacity of treatment wetlands. Removal or assimilation of various contaminants can be affected by the hydraulic retention time, the timing and distribution of flows, water depths, and the sequence of shallow vegetated areas versus deeper open water areas.

Hydraulic retention times determine the duration of settling and contact between contaminants of concerns and the sites where treatment is occurring. These "sites" can include the biofilm of bacteria and fungi attached to the surfaces of plants, on dissolved or suspended organic matter, and in wetland sediments. Longer retention time also provides increased opportunity for uptake or sequestration of contaminants by wetland sediments, plants, and fauna (Wetzel 2001). Hydraulic retention time can be increased either by increasing the area of the wetland, routing water in a sinuous pattern through the wetland, by increasing the water depth, or increasing the "drawdown" time (Hammer and Kadlec 1983). Wetland size is often constrained by adjacent land uses, and the effect that increasing water depth has on the various treatment mechanisms within the wetland places a practical limit on the extent that this hydraulic retention time can be maximized (Kadlec and Knight 1996). However, given a wetland of a certain size and water depth, hydraulic retention time can be maximized by assuring an even distribution of flow throughout the entire wetland and minimizing short-circuiting (where the passage of some flow in the wetland results in a less-than-expected residence time) and the existence of dead spaces where water is not well mixed (Kadlec 1989).

Optimizing the ability of a wetland to remove a suite of constituents requires a variety of water depths and velocities within the system. Overland flow through vegetation or movement of water into deeper open water ponds causes velocities to decrease, promoting the flocculation and settling of suspended matter that is often bound with heavy and trace metals, phosphorus, and pathogens (Delaune and Gambrell 1996). Deepwater areas promote the development of anaerobic conditions in wetland sediments, which is essential for sequestration of heavy metals. Anaerobic areas are also critical for denitrification (conversion of nitrate to nitrogen gas), which is the most important mechanism for the permanent removal of nitrogen (Reddy et al. 1989). Conversely, shallow, vegetated zones or areas of water trickling over gravel or boulders promotes increased contact with sites of biogeochemical activity and enhances oxygenation of surface waters, thereby promoting aerobic treatment processes such as inorganic phosphorus sequestration, nitrification (conversion of ammonia to nitrate), precipitation, and biological uptake of some contaminants (Patrick 1992; Reddy et al. 1989).

Consistent inflow to a treatment wetland is also an important factor in achieving predictable treatment efficiencies. Particularly in wetlands constructed to treat urban runoff, the pulsed flow of water into the wetland can be minimized by providing for a sediment forebay (a deepwater pond or concrete box) designed to handle high discharge rates and provide for maintenance access (Kadlec and Knight 1996). In addition to dampening peak flows and reducing water velocities, forebays provide the first opportunity to settle contaminants associated with the suspended load and minimize scouring of sediment and vegetation in the wetland thereby reducing transport of contaminants to downstream areas.

Because optimization of the treatment capacity of a wetland requires a variety of physical and hydrologic conditions, some constructed treatment wetlands have several different zones, each of which is designed to enhance a particular removal mechanism (Kadlec and Knight 1996). These zones may include a sediment forebay and alternating deep-water areas and shallow vegetated areas to maximize aerobic and anaerobic treatment processes.

Manipulating Wetland Biota

The biota within a wetland also affect its treatment capacity (as well as its suitability for specific wildlife). Vascular plants are the facet of wetland biota most commonly manipulated to enhance treatment goals (Kadlec and Knight 1996). The various species, guilds, and growth forms available (e.g., annual or perennial, emergent, floating or submerged, woody or herbaceous) each fill different niches within the wetland and affect the treatment capacity of the wetland (Wolverton and McDonald 1987).

Vascular plants contribute to pollutant assimilation/treatment capacity in a variety of ways: 1) they accumulate, stabilize, and transform contaminants, 2) they modify the physical and chemical properties of soil, 3) they provide organic carbon substrates to adsorb contaminants, 4) they support microbial respiration through the release of root exudates and decomposition products, 5) they improve aeration of the root zone and increase porosity of the upper soil zones, 6) they intercept and retard movement of chemicals, and 7) they decrease vertical and lateral migration of pollutants to groundwater by extracting available water and reversing the hydraulic gradient (Susarla et al. 2002). Precise treatment mechanisms vary based on the species, mineral composition, and nutrition requirements of the plant (Boyd 1978). Floating, submerged, and emergent vascular plants have been shown to store metals (Klumpp et al. 2002; Mcgrath et al. 2002), nutrients (Reddy and Debusk 1987), and priority organics (Susarla et al. 2002) at different rates, with most storage for emergent macrophytes occurring below ground in the roots (Taylor and Crowder 1983). However, it should be noted that storage of contaminants in vascular plants is minor compared to sequestration in sediments. Maximum uptake of nutrients and contaminants typically occur during the early stages of plant growth and decreases as the plant matures. As plants senesce, they drop their leaves and decompose thereby releasing nutrients and organic matter back into the system (Whigham et al. 1989). Floating vascular plants, such as duckweed (*Lemna* spp.) and water hyacinth (*Eichhornia crassipes*), can be maintained at an exponential growth stage in order to maximize pollutant removal rates (Reddy and Debusk 1987). However, plant biomass must be harvested frequently in order to maintain these high growth rates, a practice that often proves impractical because of ecological consideration, cost and logistics. Ultimately, the choice of which species of vascular plants to use in a treatment wetland is dictated by the desired treatment, the hydrologic regime of the wetland, conditions that affect growth, survival, and reproduction (e.g., nutrient levels, flooding, DO, pH), and any ancillary benefits that the vegetation may provide (e.g., habitat; Kadlec and Knight 1996).

Although they are not the dominant source of treatment for most pollutants, plants are essential for the final polishing of surface water (e.g., bringing low pollutant concentrations down to background levels; Kadlec and Knight 1996) and for providing physical structure for the attachment of bacterial biofilms, increasing the surface area available for biogeochemical treatment of contaminants (Wetzel 2001). Biological uptake via vascular plants is not a permanent removal mechanism; nutrients or contaminants stored as plant biomass will eventually be liberated to the sediment or water column as the dead tissue or litter decomposes and is remineralized (Mulholland and Kuenzler 1979). Nevertheless, the organic matter produced by vascular plants is important in supporting heterotrophic bacteria, which are important for nutrient transformation and sediment oxidation-reduction reactions. Plant roots provide a substrate for the growth of attached bacterial biofilms. They also increase the oxygen content of sediments and release exudates which cause many contaminants to precipitate, and provide for the degradation of certain compounds through enhanced bacterial and fungal activity in the rhizosphere (Susarla et al. 2002).

Enhancing Treatment Capacity Through Management and Maintenance

The assimilative capacity of treatment wetlands can be further enhanced through periodic management and maintenance. These activities include: 1) the regulation of influent flow rate and wetland water levels to keep hydraulic and contaminant loading rates within design parameters, 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, 4) upstream source control to enhance assimilative capacity, and 5) 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).

C. Considerations for Using Wetlands to Treat Urban Runoff

While most of the published research on treatment wetlands has been conducted on wetlands designed to treat municipal or industrial wastewater, there is a growing body of literature evaluating the effectiveness of wetlands as best management practices (BMP) to treat urban runoff. More information is needed on the effectiveness of treatment wetlands as stormwater BMPs particularly in semi-arid or arid environments (Strecker et al. 2001). There is currently an effort to collect and evaluate standardized performance data for stormwater BMPs through the National Stormwater BMP database (www.bmpdatabase.org). The USEPA-funded database, developed by a team of stormwater experts associated with the Urban Water Resources Research Council of the American Society of Civil Engineers, is one component of a broader project with the ultimate purposes of identifying factors that affect BMP performance, developing measures for assessing BMP performance and using the findings to implement design improvements (Clary et al. 2002). There are also regional efforts underway to study the effectiveness of treatment wetlands as urban runoff BMPs through the Stormwater

Monitoring Coalition (www.sccwrp.org/about/rspln2002-2003.html#stormwater monitoring).

In general, the design of treatment wetlands as urban runoff BMPs must take into account the highly variable hydrology and contaminant loading typical of storm and dry weather flows. In addition, concentrations of pollutants in storm water and dry weather runoff can sometimes be much higher than in wastewater, depending on the land use and contaminant of concern (Table 1). Therefore, urban runoff treatment wetlands must be designed to accommodate this variability.

In order to appropriately design treatment wetlands in southern California, a better understanding is needed of the pollutant loadings in urban runoff from various land use types in semi-arid and arid climates (Schiff and Sutula 2002). In addition, it is important that such studies must distinguish stormwater from dry season 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.

TABLE 1. Comparison of Pollutant Concentrations in Municipal Effluent to Stormwater (all concentrations in mg L^{-1} except for metal data in μg L^{-1})

Pollutant	Municipal Effluent [a,c]	Stormwater Event Mean Conc. (Natl Ave)[b]
TSS	47.0	78.4
BOD	85.0	14.1
COD	121.0	52.8
Total N	37.0	2.39
Total P	2.3	0.32
Soluble P	NR	0.13
Copper	35.0	14.0
Lead	0.67	68.0
Zinc	47.0	162.0

References: [a] SCCWRP, unpublished data
[b] Caraco (2000)
[c] Metcalf and Eddy (1993)

While removal efficiencies of wetlands treating urban runoff have been documented, there is question to what extent these data are applicable to semi-arid and arid regions such as southern California. Runoff from semi-arid and arid regions generally have higher concentrations of contaminants than that from mesic or humid areas (Table 2, Caraco 2000). The reasons for this are as follows: 1) 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, 2) drier climates typically have higher sediment and organic carbon loading from open areas because sparser vegetative cover translates to less retention within the watershed, and 3) semi-arid climates typically have a larger average rainfall event. As a result, stormwater management practices in semi-arid regions such as southern California will likely require greater either greater pretreatment or more conservative sizing requirements than those in more humid climates (Caraco 2000).

TABLE 1. Stormwater Pollutant Event Mean Concentrations in Arid and Semi-arid Regions (all units in mg L^{-1} except metals in $\mu g\ L^{-1}$). Adapted from Caraco (2000) .

Pollutant	National	Phoenix, AZ	Boise, ID	Denver, CO	San Jose, CA
Rainfall	25	7.1 inches	12 inches	13 inches	14 inches
Sample size	2-3000	40	15	35	67
TSS	78.4	227.00	116.00	384.00	258.00
BOD	14.1	109.00	89.00	ND	12.30
COD	52.8	239.00	261.00	227.00	ND
Total N	2.39	3.26	4.13	4.80	ND
Total P	0.32	0.41	0.75	0.80	0.83
Soluble P	0.13	0.17	0.47	ND	ND
Copper	14.0	47.00	34.00	60.00	58.00
Lead	68.0	72.00	46.00	250.00	105.00
Zinc	162.0	204.00	342.00	350.00	500.00

Stormwater treatment systems are generally designed to reduce or attenuate peak flows as well as to control water quality. Consequently, many stormwater wetland treatment systems consist of a combination of a pond or basin designed to detain flow and trap suspended matter, and a vegetated wetland designed to remove contaminants (Breen et al. 1994). These pond/wetland systems can either be hydrologically connected to the stream channel, or located outside or disconnected from the stream channel. Carapeto and Purchase (2000) note that detention ponds or wetlands hydrologically connected to the stream channel can be much more problematic in terms on controlling the hydrologic regime and the biologic community. In addition, unlike wastewater treatment systems, management of stormwater treatment systems must address issues such as (Shutes et al. 1997):

- Pulsed and intermittent flow, that subjects wetlands to extreme fluctuations in water levels, such as sudden inundation followed by periods of completed desiccation;
- Possible resuspension of contaminants sequestered in sediments during storm events; and
- Alternative oxic and anoxic conditions that can result in remobilization of sediment-bound contaminants during the dry season.

As noted earlier in this chapter, wetland performance is generally a function of hydraulic loading rates and residence times. In stormwater treatment wetlands, these attributes are generally a function of storm intensity runoff volume, outlet structure design, and wetland size (area and volume, Carleton et al. 2001). Because of the stochastic nature of storms, sizing requirements for stormwater wetlands should be more conservative than for wastewater treatment wetlands. Unfortunately, unlike applications for wastewater, there is little design guidance for stormwater wetlands, and an absence of comprehensive long-term mass balance data necessary to evaluate the comparative performance of various design options (Carleton et al. 2001). The first sizing requirements for stormwater treatment wetlands was published by the state of Maryland (MDE 1987), which gave as a rough rule of thumb that wetland area should be at least 3% of contributing watershed area (note that area ratios are extremely simplistic and used only as a rough indication of how much land treatment make require). Strecker et al. (1992) note that an area ratio may not be as important in determining performance as volume ratio (ratio of average runoff to storage volume. Schueler (1992) recommends minimum of 2% of the contributing watershed area and treatment volume large enough to capture 90% of all storm events.

It is important to recognize that design guidelines are rules of thumb based on data largely from more humid climates, and there is still not really enough understood about design and performance of urban runoff wetlands in semi-arid and arid climates to follow these recommendations without careful consideration and further research. In a survey of stormwater management agencies in the southwestern United States, Caraco (2000) found that that sediment clogging and deposition was a major design and maintenance problem for all stormwater structural BMPs including wetlands. In addition, the unpredictability of dry season flows in arid climates may prove challenging to maintaining a stable wetland plant community. Given this fact, engineers conceptualizing designs for wetlands must have adequate information about how climate, precipitation patterns, and watershed land use will result in a range of hydrology, contaminant concentration and loading rates for a potential site. Basic stormwater and lowflow treatment wetland design guidelines must be carefully considered and adapted to local conditions. In particular, there is a need of long-term performance data on effluent quality during wet and dry season flows, and better understanding of major factors controlling appropriate sizing or dimensions data from some existing treatment wetlands such as the Prado Wetland (www.ocwd.com) and San Joachim marsh (www.irwd.com) can provide preliminary datasets to generate more in-depth questions and investigations.

D. Summary

This chapter summarized the fundamental processes that affect the capacity of a wetland to assimilate pollutants, approaches to optimize the treatment capacity of wetlands, and provides considerations for using wetlands to treat urban runoff (e.g., stormwater and dry weather runoff). Specific conclusions include:

- The cycling of compounds in wetlands in controlled by a complex series of processes that include: 1) the flocculation, settling and filtration of suspended matter, 2) adsorption and desorption of compounds from particles, 3) chemical dissolution and precipitation between waterborne solutes and solids, 4) chemical transformations, 5) volatilization and 6) biological uptake, transformation and release.

- The degree to which specific pollutant removal processes occur in wetlands is dependent on inflow concentration, the wetland hydrologic and hydraulic characteristics, the geology and chemistry of sediments, the composition of the microbial community, the flora and fauna that inhabit the wetlands, and the geomorphology and position of the wetland in the landscape.

  - The hydrologic regime of a wetland (i.e., the hydraulic residence times, depth, and frequency of flooding) influences its treatment capacity by affecting the duration of interaction between the water column and the sediments/biota; the proximity of contaminants to the sites of biogeochemical activity where treatment occurs; and the oxygen levels in the sediment and water columns.

- Characteristics of wetland sediment that influence the overall treatment capacity of a wetland include; the extent and distribution of aerobic and anaerobic zones, the quality and quantity of mineral and organic matter, and the particle size distribution.

- Many of the chemical transformations that occur in wetlands are mediated by microbial organisms. Microbial decomposition rates are highest along the aerobic/anaerobic boundaries, such as the sediment-water interface or across the surface of dissolved organic matter. Therefore, wetlands with more dissolved organic matter or more complex sediment beds provide more surface area for microbial processes to occur.

- Wetland algae and macrophytes, in their cycles of growth and decay, uptake, sequester, transform, and release compounds. When they die, macrophytes provide organic matter to the system, which increases the cation exchange capacity of the sediments and is critical to fueling microbial transformation processes.

- Vascular plants contribute to pollutant assimilation/treatment capacity in a variety of ways: 1) they accumulate, stabilize, and transform contaminants, 2) they modify soil physical and chemical properties 3) they provide organic carbon substrates to adsorb contaminants, 4) they support microbial respiration, 5) they improve aeration and increase porosity of the upper soil zones, 6) they intercept and retard movement of chemicals, 7) they provide substrate for the attachment of bacterial biofilms, and 8) they decrease vertical and lateral migration of pollutants to groundwater.

- The capacity of a particular wetland to assimilate pollutants will vary based on its intrinsic properties, such as its hydrologic, sediment, and biotic characteristics (as discussed above), and the watershed conditions, such as climate, loading rate (e.g., land use), and the storm characteristics.

- In general, the influent hydrology and the contaminant loading rates in urban runoff are much more variable and subject to periodic "spikes" that can be much higher than concentrations typically observed in wastewater. Therefore, urban runoff treatment wetlands must be designed to accommodate this variability.

- Basic stormwater treatment wetland design guidelines must be carefully considered and adapted to constraints of semi-arid and arid climates. In particular, there is a need of long-term performance data on effluent quality during wet and dry season flows, and better understanding of major factors controlling appropriate sizing or dimensions.

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CHAPTER 3.
POTENTIAL EFFECTS OF URBAN RUNOFF ON PHYSICAL PROPERTIES OF WETLAND HABITAT

A. Introduction

Stormwater and dry season urban runoff have the potential to affect the condition of wetland habitat by altering the hydrology, geomorphology, sediment transport, and water chemistry. These in turn affect the quality of the habitat available to wildlife. In addition, routine maintenance and management activities may also affect habitat within treatment wetlands. Despite the large body of literature on the effectiveness of treatment wetlands, there has been relatively little effort made to study and summarize how their use in water treatment affects habitat functions. While the use of treatment wetlands by wildlife has been documented, the general literature contains no systematic surveys of the quality of habitat found in treatment wetlands, particularly for those used to treat urban runoff (Knight et al. 2001; Wren et al. 1997). In the literature we found only one study which quantified the habitat value of constructed wetlands used to treat municipal discharge (McCallister 1992). Azous and Horner (2000) co-authored a rigorous, comprehensive study of the effects of urbanization on natural wetlands in the Seattle, Washington area. Much of this work is applicable to southern California wetland ecosystems and provides an excellent framework to approach the assessment of habitat value. However, the physical and biological attributes of wetlands in semi-arid and arid climates such as in southern California can differ greatly from those found in temperate, humid climates like the Pacific Northwest, so the relevance of this study to southern California wetlands has limitations.

In considering the potential effects of treatment wetlands on wildlife, there are several key questions:

1. Do wetlands created or enhanced for water quality improvement of urban runoff become environmental hazards for wildlife?

2. How does directing urban runoff or stormwater through or a natural or restored wetland affect the habitat of the wetland?

3. How do considerations of the habitat effects of treatment wetlands differ by wetland class (e.g., how are the issues different for riverine vs. palustrine or lacustrine wetlands)?

Utility of the Risk Assessment Paradigm in Assessing the Habitat Value of Treatment Wetlands

Wren et al. (1997) and Lemly and Olendorf (2002) suggest that a risk assessment framework is a useful paradigm to examine the potential hazards associated with presence of contaminants in constructed wetlands (Figure 2; Lewis et al. 1999). Proper application of this approach can reveal potential problems and form the foundation for the selection of an ecologically-sound treatment option (Lemly and Ohlendorf 2002). A typical ecological risk assessment consists of three steps: 1) problem formulation, 2) analysis, and 3) risk characterization (USEPA 1998).

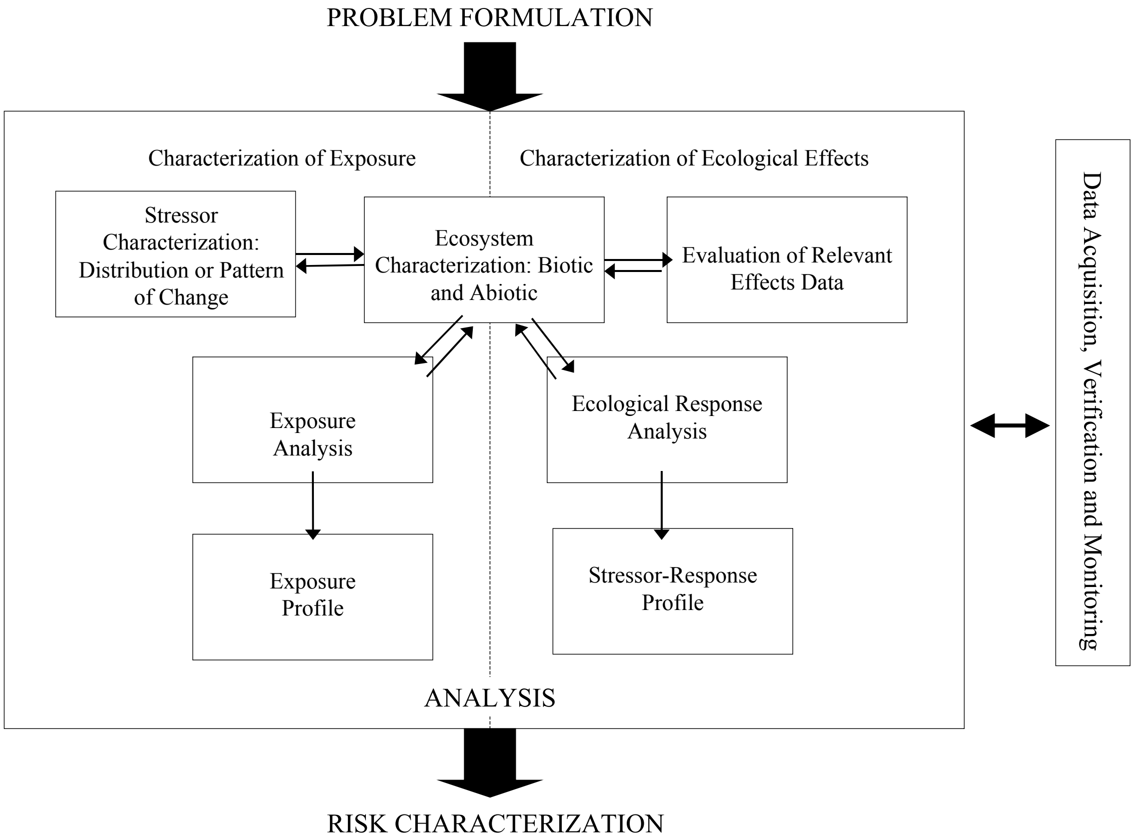

FIGURE 2. Schematic of ecological risk characterization

Problem formulation begins with the identification of ecological objectives or endpoints, which are typically the resources to be protected (e.g. maintain integrity of wetland food chain that provides habitat for fish). Management goals, policy drivers, resource needs are important in identifying these ecological endpoints. For each assessment endpoint, measurement endpoints (one or more parameters that can be quantified) are selected through a wetland characterization (baseline physical and biological survey) and stressor and receptor identification. An analysis plan is developed to guide the next phases of the assessment. The importance of the problem formulation phase step cannot be understated. Shortcomings consistently identified in case studies include 1) the absence of clearly defined goals, 2) endpoints that are ambiguous or difficult to define and measure, and 3) failure to identify important risks (USEPA 1993, 1994)

The analysis step of the risk assessment will have two components: 1) an exposure assessment, in which the level of contamination or damage to the habitat is quantified, and 2) ecological effects assessment, in which the effect of the damage or contamination to a particular species is assessed (Figure 2). Once the analysis step is completed, the results of the findings are integrated and interpreted in context of risk to fish and wildlife through a risk characterization.

Risk assessments are best conducted through a phased approach, in which work completed in one phase helps to determine whether further studies are required (Lemly and Ohlendorf 2002).

We have utilize the risk assessment paradigm to organize the concepts presented in the literature review. Chapter 3 considers potential effects on the physical, chemical, or biological condition of wetlands used to treat urban runoff, while Chapter 4 considers how these potential changes in treatment wetlands may affect wildlife (e.g., do they pose a risk). In addition, the phased approached of the risk assessment is also reflected in the literature recommendations for monitoring presented in Chapter 5.

It is important to note, that although the potential for treatment wetlands to threaten wildlife has been identified as a concern (Helfield and Diamond 1997), there is very little in the general literature that directly addresses the above questions. Consequently, we have drawn on a variety of related literature sources to extrapolate general conclusions and to help provide insight into potential effects and key considerations.

B. Effects on Wetland Hydrology

Hydrology is the primary driving force controlling wetland structure and function (Mitsch and Gosselink 1993). The interaction of hydrologic processes, geomorphology, landscape position, and sediment yield to the system control the physical structure of the channel, banks, and floodplain, and elevation of wetland sediments. This in turn affects the habitat within the wetland. Therefore, changes in hydrology can result in change in the nature or quality of wetland habitat.

Increased urban runoff has been shown to result in changes in the magnitude, frequency, and timing of flow to wetlands. These changes have been best documented in riverine wetlands, which are immediately impacted by changes in watershed hydrology.
Increases in urban runoff are generally proportional to increases in effective impervious surface (e.g., impervious surface that is hydraulically connected to a drainage system) and associated changes to the nature of the drainage network. These changes result in a decrease in infiltration and an increase in surface runoff (Dunne and Leopold 1978). This shift in hydrology has the following documented effects on riverine wetlands and riparian areas, and would have similar effects in palustrine and estuarine wetlands:

- *Changes in dry season flow patterns*
  The precise effect will vary depending on the geology of the watershed and the amount of impervious surface. In watersheds with groundwater derived baseflow, increases in impervious surface can decrease infiltration and ground water recharge, thereby reducing the period of the season in which baseflow is present (Barringer et al. 1994; Espey et al. 1965). However, particularly in arid climates, dry season flow may increase in watersheds with extensive urbanization (and/or

where groundwater flow is a minor contribution to baseflow). For example, Hamilton (1992) documented an exponential increase in dry season flow of several streams in southern California and noted that "perennialization" of intermittent or ephemeral streams is typically observed in watersheds with greater than 40% urbanization. Much of this flow is attributed to irrigation of lawns and landscapes areas at rates ranging from 40-60 inches per year.

- *Increases in magnitude and duration of peak flows*
  Peak flows increase with increasing impervious surface in the watershed. The relative magnitude of increased peak flow is greatest for high frequency events (i.e., 1-5 year events), which can increase up to 20-fold, and less dramatic for extreme events (i.e., 50-100 year events, Hollis 1975). Booth (1990) reported a 30 fold increase in the duration of stream flow during a 2-year storm event in catchments with 20% impervious surface. Similarly total runoff volume increases by up to 40%. Other studies have reported much higher increases in total runoff volume ranging from 100-1000%. Changes in the magnitude of peak flows can result in increased <u>frequency of channel-forming events</u> (i.e., the frequency of runoff events with sufficient stream power to mobilize bedload sediments and cause changes in the channel, Bledsoe and Watson 2001). Similarly MacRae (1992) observed that increased impervious surface resulted in increases in the frequency of moderate flow events that (e.g., geomorphically significant flows) that result in changes to the structure of the stream bed and bank. An increase in channel forming events will result in the scouring or excess sedimentation of channel and bank habitats.

Increases in urban runoff and/or diversion of storm flows have been show to have dramatic impacts on natural palustrine wetlands (Azous and Horner 2000). Diversion of stormwater or urban runoff to natural or created wetlands (including wetlands within stormwater detention ponds), may result in larger, more frequent, and more rapid water level fluctuations (Azous and Horner 2000). Increases in sudden inundation of wetlands can cause stress on the vegetation and fauna (Azous and Horner 2000). Azous et al. (2000) showed that hydrologic changes in palustrine wetlands associated with increasing urbanization had more immediate and measurable effects on composition of vegetation and amphibian communities than other environmental conditions. Because of alterations in natural drainages and the reduction in groundwater recharge from increased impervious surface, these wetlands are also more subject to periodic drawdowns in water level and desiccation.

Impacts of increased urbanization on the hydrology estuarine wetlands have also been fairly well documented, though literature is lacking for coastal southern California. As a result of increased development and disturbance in the watershed, sedimentation rates in coastal wetlands have substantially increased. Onuf (1987) showed a 40% decrease in the tidal prism of Mugu Lagoon due to agricultural-related increases in sediment input. Calhoon et al. (1996) measured extremely high rates of sediment accretion (up to 8 cm yr^{-1}) in Tijuana Estuary from storms in the early 1990s. These rates are much greater than the long-term average and have resulted in major habitat conversion within the estuary (Mudie and Byrne 1980; Weis et al. 2001). Changes in watershed hydrology and sediment transport are exacerbated by modifications in the land use surrounding estuarine wetlands. Reduction or elimination of upland buffers between wetlands and urban land uses, and the diversion of storms flows into these wetlands have resulted in changes in salinity regime and patterns of flow, sedimentation and erosion within the estuary (Nordby and Zedler 1991).

C. Effects on Geomorphology and Sediment Transport

Wetlands may experience geomorphic alterations associated with changes in the amount or nature of sediment input to the wetland. Changes in the nature or magnitude of sediment loads may result in aggradation or degradation of wetlands as they adjust to accommodate their new sediment loads. Potential geomorphic effects in treatment wetlands include:

- *Aggradation* – Diversion of runoff with a high sediment load to a stream or wetland will generally cause the system to aggrade (i.e., increase in the elevation). Stormwater runoff from areas under development (i.e., construction sites) typically have high sediment loads. A sediment-overwhelmed stream will have a tendency to increase shear stress on the banks from flows being diverted around deposits, ultimately resulting in channel widening and increased overbank deposition, or downgrading of the channel and erosion of the banks (Paul and Meyer 2001). High sediment deposition rates in estuaries can lead to major loss of wetland habitat (Mudie and Byrne 1980; Weis et al. 2001).

- *Degradation or entrenchment* – Runoff from landscapes that have already been "built out" or developed generally conveys less sediment than runoff from natural areas. Decreases in the sediment supply to a stream or wetland will tend to destabilize the channel by incising its bed, leading to bank failures. This pattern of channel deepening (and eventual widening) may lead to erosion of sediments from channels, flats, banks, and bars, remobilization of contaminants sequestered in these sediments and degradation of vegetation and biota (Azous and Horner 2000; Booth and Jackson 1997; Booth 1990). The phenomenon is occurring in the San Diego Creek watershed were it is estimated that >95% of the sediment load to Upper Newport Bay is from channel erosion (NTS Master Plan 2003). Loss of wetland vegetation can further destabilize the channel, exacerbating incision. Furthermore, incised channels may result in a lowering of the local groundwater table in the adjacent floodplain and result in a reduction of the extent of wetland or riparian habitat.

- *Changes in sediment grain size distribution* – The sediment load, its particle size range, input timing and mechanisms, and longitudinal distribution contribute to the development of geomorphic surfaces and deposits that form the foundation for riparian and aquatic habitat. (Schueler 2000a). Diversion of stormwater or dry season flow to a wetland or riparian area can alter the particle size distribution of sediments found in the wetland, due to either change in sediment supply or changes in hydrodynamic sorting within the channel or basin (Paul and Meyer 2001). For example, the construction of stormwater ponds or sediment detention basins can greatly alter flow and sediment dynamics in riverine wetlands and riparian areas; these features are excellent at trapping coarser-grained gravels and cobbles, but not as efficient at retaining silts and clays (Schueler and Lugbill 1990). Where constructed wet ponds or dry sediment detention ponds have been placed within stream drainage networks, these features interrupt the hydrological and sediment transport processes occurring on landscape scales (Paul and Meyer 2001; Petts 1984). In particular, they act as a barrier to the downstream movement of bedload, but export fine particulates. This can result in the embedding of downstream substrates with these fine-grained sediments, resulting in reduction of habitat value (Galli 1988). These changes can affect many ecological processes, from filter-feeding organisms (Hart and Finelli 1999) to carbon and nutrient cycling (Jones and Mulholland 2000).

- *Changes in topographic complexity* – Reductions in topographic complexity eliminate structural elements that are important in the biogeochemistry and ecology of these systems (Finkenbine et al. 2000). Higher flow velocities typically associated with stormwater can scour and homogenize channel beds and reduce the amount of large woody debris that is retained in riverine wetlands. Constructed wetlands normally lack the complex microtopography found in natural wetlands. This can be due to the 1) difficulty and associated cost of creating microtopography with construction equipment and the necessity of providing for easy access for maintenance and 2) the necessity of designing the wetland which optimizes flow conditions for pollutant removal (e.g., minimizing dead spaces, etc., Kadlec and Knight 1996).

It is important to note that the sensitivity of a particular wetland to changes in geomorphic processes will vary based on the nature and setting of the wetland. For example, increasing flow to a stream with coarse, embedded substrate will likely result in moderate widening of the banks. In contrast, increasing flow to a sandy stream will likely result in channel incision, followed by bank collapse. The effects of urbanization on geomorphic processes (and therefore habitat) can vary both spatially and temporally. For example, during the initial build-out of a catchment, sediment loads can temporarily increase, especially in the absence of well-designed and maintained construction BMPs, resulting in stream aggradation. However, upon completion of urbanization, the total sediment yield often decreases dramatically while runoff remains high. Furthermore, in many areas, post-urbanization sediment yield may consist of a higher proportion of fine-grained material. This shift can affect the suitability of wetlands to support specific flora or fauna (e.g., bury spawning beds, asphyxiate juvenile fish or amphibians) and as well as affect the assimilative capacity of the wetland sediment (see next section).

When considering the effects of urbanization and urban runoff on wetlands and riparian areas in semi-arid climates such as Southern California, it is important to consider the highly "pulsed" nature of these ecosystems. In particular, episodic "pulsing" events, such as El Niño, extreme floods, fire, and debris flows are believed to be the dominant controlling factors in defining the characteristics of many stream channels in Southern California (Hecht 1993). These intermittent but natural occurring events temporarily establish a new set of stream conditions and vegetation patterns that gradually returns to similar hydraulic geometry over time. Pulsing events of this nature have also been shown to be extremely important in controlling wetland productivity, as they are responsible for periodic transport of fresh sources of carbon and nutrients to the wetland (Conner et al. 1989). Disruption, increase, or elimination of these episodic patterns can result in the most catastrophic effects on the long-term viability of wetland and stream systems. However, these events are precisely the ones that stormwater management agencies strive to control; this issue can be one of the most difficult challenges to address.

D. Effects on Wetland Sediment and Surface Water Chemistry

The majority of published monitoring studies involving treatment wetlands over the last 20 years have monitored influent and effluent parameters, but have not typically measured changes in water quality or sediment parameters within the wetland itself (Horner et al. 2000; Kadlec and Knight 1996). In response to this criticism, studies are now being published with more comprehensive data, but overall information is still relatively limited, particularly for urban runoff treatment wetlands in arid and semiarid climates. However, extensive research exists on effects of urbanization on receiving waters such as streams, estuaries, and some natural wetlands (USEPA 2000b). This research can be drawn upon to make some initial inferences about effects of urban runoff (wet and dry weather) on natural and constructed wetlands.

The ability of natural or constructed wetlands to positively affect sediment and surface water chemistry is precisely the reason they have gained attention as water quality management tools. The degree to which these chemical changes occur (and the potential effect of these changes on habitat within the wetland) depends on a variety of spatial and temporal factors that control physical and biological contaminant cycling. These factors include landscape position, hydrologic regime, design or morphology of the constructed or natural wetland, hydraulic and contaminant loading rates, rates of internal contaminant cycling within the wetland, and a range of other site-specific factors.

Water Temperature and Salinity

The potential effects of urban runoff on the salinity of streams or wetlands will vary depending on the conductivity of the runoff and the baseline conditions in the receiving stream or wetland. Urban runoff is often moderately saline, especially if it includes reclaimed water or is draining areas with some degree of agriculture. For example, Horner et al. (2000) found increased conductivity in wetlands with highly urbanized catchments; however, there has been no comprehensive study of the impact of salinity from urban runoff on freshwater aquatic ecosystems. Alternatively, intermittent streams in coastal California may naturally be moderately saline due to base flow leaching of minerals from saline or alkaline geologic parent material. In this circumstance, the increased volume and duration of freshwater input from urban runoff may result in a shift in plant community composition if salinity levels are different from background conditions (Day 1989; Wetzel 1983). Furthermore, increased freshwater input to estuaries (via coastal streams) during the dry season can have an impact on the typical salinity regime and the native biota adapted to live in that environment. Kuhn and Zedler (1997) found a decrease in native plant species diversity and an increase in exotic plants associated with increased freshwater input from urban sources, while Nordby and Zedler (1991) found major impacts to benthic infauna, macroinvertebrates, and fish communities associated with impoundment of urban runoff in an estuarine lagoon with a closed ocean inlet during the dry season.

Urban runoff and upstream channel alterations can also affect stream or wetland biota by altering the water temperature, which is one of the major factors that influence rates of major chemical and biological processes (Wetzel 1983). Increased dry season or stormwater runoff may result in stream destabilization, incision, and subsequent loss of streamside vegetation due to bank cavitation. The loss of canopy cover can increase stream temperature, which can affect the rate of detrital processing, respiration, bacterial growth, as well as timing of reproduction of reproduction, molting and drift of aquatic organisms (Wetzel 1983). Changes in water temperature have also been associated with algal blooms, alterations in macroinvertebrate species composition, and decline in salmonid species (Paul and Meyer 2001). Wetlands and riparian areas that lack significant shading from vascular plants (e.g., wet ponds or dry sediment detention ponds) can act as a heat sink and discharge warmer waters during storm and baseflow periods (Galli 1991). Alternatively, increased urban runoff can promote growth of in-stream riparian vegetation, which may increase canopy cover and reduce stream temperature. In this circumstance the opposite temperature effects may be observed.

Nitrogen, Phosphorus, Organic Matter and Dissolved Oxygen

The design and management of treatment wetlands to remove nitrogen, phosphorus, and organic carbon in wastewater, industrial, and nonpoint pollution (particularly dry weather flows) is well advanced. There are extensive literature reviews and textbooks on nitrogen (N), phosphorus (P), and carbon (C) cycling in both natural and constructed treatment wetlands (Hammer 1989; Johnston 1991; Kadlec and Knight 1996; Moshiri 1993). An extensive body of literature also exists on the impact of nutrient enrichment and excessive organic carbon loading, known as eutrophication, on aquatic ecosystems (Cooper and Lipe 1992; Hillbricht-Ilkowska 1993; Malone and Conley 1996; Turner and Rabalais 1991).

Sources of nutrients and organic matter in urban catchments include fertilizers from residential and agricultural runoff, recreational and commercial animal husbandry (e.g., horse farms, cattle lots, poultry and swine production), atmospheric deposition, and many industrial and commercial operations (Heany and Huber 1984). High loading in both wet and dry weather flows can be further exacerbated by occasional failures in the sanitary sewer system operations, resulting in sewage spills. The variability in land use and thus loading from the watershed, in addition to the stochastic nature of storm flows, make estimation of nutrient and organic carbon loading to wetlands via urban runoff difficult.

The loading of nutrients and organic carbon to a wetland controls the biological productivity of primary producers such as algae and plants and the rates of heterotrophic bacterial production and respiration (Odum 1971). This biological activity exerts major controls on the biogeochemical processes occurring in both surface waters and sediments, including control of surface water and sediment pH, dissolved oxygen content and sediment oxidation-reduction potential (Delaune and Patrick 1980; Gambrell and Patrick 1978; Patrick 1992). These parameters in turn control the cycling of a host of other contaminants including trace and heavy metals, organic contaminants, and bacteria (as is described below).

Assimilation of organic carbon, nitrogen and phosphorus in natural and constructed wetlands occurs through processes such as: 1) decomposition (through respiration, fermentation, methanogenesis, reduction reactions such as sulfate, nitrate, and iron reduction), 2) biological uptake, 3) sedimentation and sequestration, and 4) nitrification and denitrification (Mitsch and Gosselink 1993). Of these processes, permanent removal mechanisms only exist for nitrogen (denitrification) and carbon (methanogenesis, respiration). Phosphorus is typically stored in sediments, and to a lesser extent as plant biomass, but can be remobilized to the surface waters during anoxic conditions in sediments, erosion of sediments during storm events, or processes of death and decomposition of plant biomass (Patrick 1992). Anthropogenic sources of nutrients in wastewater and nonpoint source pollution, such as agricultural and urban runoff, are typically in the dissolved inorganic state (nitrate, nitrite, ammonia, phosphate) – a form that is preferred by plants and algae for uptake (Mitsch and Gosselink 1993). For both nitrogen and phosphorus, one of the most important treatment processes that occur in wetlands is the conversion of the dissolved inorganic forms to organic forms, which take longer to decompose and recycle back to dissolved inorganic forms and therefore are considered less of a nuisance from a water quality perspective.

Extensive literature supports the idea that under *conservative* loading rates, many types of natural and constructed wetlands can sustainably assimilate nitrogen, phosphorus and organic carbon inputs into the system (Kadlec and Knight 1996; Mitsch and Gosselink 1993). In southern California, the most extensive data on the performance of treatment wetlands is available for nitrogen and phosphorus (see NTS Master Plan 2003 for summary) The assimilative capacity is contingent on a number of factors described in Chapter 2 of this review; therefore, not all wetlands are equivalent in their capacity to treat nutrient and organic carbon loading. If overloading occurs, the effects can be multiplicative, including excessive algal growth, large fluctuations in pH and chronically low dissolved oxygen conditions in surface waters, low redox potential (anoxic) sediments, and subsequent release of heavy metals, phosphorus and organic contaminants sequestered there (Mitsch and Gosselink 1993). High nutrient concentrations can also result in shifts in species composition of algal and plant communities toward species with higher nutrient tolerance (McCormick et al. 1996; McCormick and O'Dell 1996). Furthermore, the structure of the benthic community can change dramatically with organic matter enrichment of sediments (Mann 1982; Pearson and Rosenberg 1978). Therefore, if treatment wetlands are to be used to manage urban runoff, more research is needed in the long-term effects of nutrient loading on wetland biota. Research is also needed in the design and management of constructed wetlands to optimize sustainable nutrient removal, and to understand the implications of these engineering options on the ecology and habitat value of the wetland (Knight et al. 2001).

Heavy and Trace Metals

Accumulation of heavy and trace metals from urban wet and dry weather runoff is one of the greatest concern in terms of potential bioaccumulation and ecotoxicity in treatment wetlands (Paul and Meyer 2001; Schueler and Holland 2000b). In addition to minerals naturally enriched in heavy and trace metals, recognized sources include industrial discharge, atmospheric deposition, and wear from brake linings, tires, and engine parts (Schueler and Holland 2000b). Stormwater ponds draining highways are typically the most heavily enriched by heavy metals, followed by commercial sites, then residential areas (Schueler 2000a). In general, metal concentrations in nonpoint source runoff often exceed that found in point source discharges (USEPA 2001).

The principal mechanism for removal of heavy and trace metals in natural and constructed wetlands is sedimentation and sequestration in sediments. Because many metals have an affinity for the surface area of particles, most of the metals in urban runoff are associated with suspended sediments rather than available in a freely dissolved form in surface waters (Wilbur and Hunter 1977). Consequently, although assimilation by vegetation is an important mechanism for increasing the bioavailability of metals to wetland biota, it accounts for less than 1% of removal that occurs in treatment wetlands (James et al. 1990). Under anoxic conditions common in sediments with high organic matter and/or nutrient loading, previously sequestered metals can be remobilized from the sediments and may become bioavailable (either within the wetland or at downstream areas). Dry season conditions with higher temperatures and stagnant water generally promote anoxia in sediments. For this reason, Schueler (2000a) reports that metal removal efficiencies in wetlands treating urban runoff are generally higher during storm flow than dry season flows. Other authors report that palustrine wetlands and stormwater ponds are often sources of metals during dry periods (Kadlec and Knight 1996; Mungur et al. 1999).

Diversion of dry or wet season urban runoff to treatment wetlands can also affect concentrations of metals in the sediments by influencing the hydrodynamic sorting of sediments, which is one of the major factors responsible for variations in heavy and trace metal concentrations found in wetland sediments (Jain and Sharma 2001; Rubio et al. 2000). Many sediment-bound pollutants tend to be associated with the fine (<2 μm in size) particle fraction, which often predominates in runoff from developed catchments (Bavor et al. 2001). Rubio et al. (2000) found that copper, zinc, lead, chromium concentrations were four to seven times higher in fine-grained sediment than in coarser sandy sediments. In addition, metals such as nickel, cobalt, selenium, and arsenic have an affinity to adsorb to iron and manganese oxyhydroxide and aluminosilicate minerals (Daskalakis and O'Conner 1995; Garcia-Hernandez et al. 2000). For this reason, treatment wetlands (natural and constructed) with a quiescent hydrologic regime and fine grain sediments are typically more heavily enriched in metals (Schueler 2000b). Horner et al. (2000) found that sediment metal concentrations generally decline with distance from the effluent.

When considering bioaccumulation of metals in treatment wetlands, it is important to recognize that natural background levels of certain metals can be quite high, depending on factors such as basin geology and mineralogy (Stein et al. 1996; Warren 1981). For example, areas underlain by Cretaceous marine sedimentary rock can have high concentrations of selenium in surficial sediments, groundwater, and surface water as a result of natural enrichment from the geologic parent material (Garcia-Hernandez et al. 2001b; Presser et al. 1994). Wetlands in these areas can sequester metals such as selenium in the sediment independent of anthropogenic inputs (Garcia-Hernandez et al. 2000; Welsh and Maughan 1994; Zhang and Moore 1996). Investigators have hypothesized that the high concentrations of selenium found in groundwater in the San Diego Creek watershed, California are due to oxidation and leaching of selenium from sediments of the historic "Swamp of Frogs," now drained and converted to urban land use (Hibbs, personal communication). In arid and semi-arid regions such as southern California, high evapotranspiration rates in wetlands can further concentrate metal loadings, and thus potentially create a concern with bioaccumulation (Schueler and Holland 2000b).

It is important to distinguish sediment accumulation from bioaccumulation and toxicity. Total sediment metal concentration does not necessarily equate to a level of bioavailability or toxicity in biota. Various studies have found that the percentage of total metal concentration susceptible to leaching and therefore potentially bioavailable is generally less than 10% (Lau and Chu 2000). While several studies have looked at the partitioning of metals into major mineral phases used to predict the potential for remobilization to surface waters, currently there is no connection describing how these major phases correlate to bioaccumulation and toxicity in biota (Hoai et al. 1998). Potential ecotoxicological risks associated with bioaccumulation are discussed in more detail in Chapter 4.

Organic Contaminants

Pesticides, hydrocarbons, and other priority organic compounds are another group of contaminants in urban runoff that has the potential to affect the habitat value of treatment wetlands. Urban uses account for more than a third of pesticide applications in the United States, where they are frequently applied around homes, commercial and industrial buildings, and are intensively used in lawn and golf course maintenance (LeVeen and Wiley 1983, USGS 1999). Areal application rates in urban environments frequently exceed those in agricultural operations by an order of magnitude, and are often less stringently regulated (Schueler 1994b). Hydrocarbons and other sources of priority organic compounds such as polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and petroleum-based aliphatic hydrocarbons are also commonly found in urban runoff, particularly from nuisances flows from industrial or commercial sites and from highways (Maltby et al. 1995a; Maltby et al. 1995b).

The major mechanisms for removal of organic contaminants in treatment wetlands include: 1) photochemical oxidation, 2) sedimentation, 3) sorption, 4) volatilization, and 5) biological (microbial) degradation (Kadlec and Knight 1996). While the chemistry of organic contaminants and partitioning in the surface waters and sediments is highly variable, a strong correlation exists between contaminant concentration and suspended sediment, suggesting that sedimentation may be a major removal mechanism for organic contaminants in wetlands (Kadlec and Knight 1996; Pereira et al. 1996). While several studies have recently been published characterizing the concentrations of organic contaminants in urban runoff, including studies conducted in semi-arid climates such as southern California (e.g., Schiff and Sutula 2002), limited data exists on the fate of these contaminants in natural and constructed wetlands. For example, partitioning of organic contaminants between the water column and the sediments, processes of transformation in wetlands, and their effect on wetland biota still require additional study (Gavens et al. 1982; Maltby et al. 1995a; Maltby et al. 1995b).

Bacteria

Bacterial loading to wetlands and streams in urban areas has been well documented as one of the most common pollutants potentially affecting aquatic systems (Porcella and Sorenson 1980; Simpson et al. 2002). Notably, coliform bacteria are used as an indicator of human pathogens, though many argue that they are so ubiquitous and not unique to human fecal contamination that their usefulness is limited. Studies of wet weather runoff from a variety of land uses in the Los Angeles River watershed, California, showed high concentrations of total and fecal coliform and E. coli from horticultural nurseries, horse stables, residential, commercial, and mixed agricultural land uses. In all cases, bacterial concentrations emitted from these sites were at least an order of magnitude higher than the recommended water quality standards (M. Sutula, unpublished data). In addition, sewage spills or leaks may contribute to high coliform counts in urban streams and downstream wetlands (Paul and Meyer 2001).

The main mechanism for removal of indicator bacteria from urban runoff in treatment wetlands is through the reduction of suspended sediment (Gersberg et al. 1989; Stenstrom and Carlander 2001). Bacteria typically aggregate to the organic matter found on fine-grained suspended and bed sediments (< 2 μm fraction). Bavor et al. (1987) concluded that wetlands that have alternating zones of shallow marsh and open water were the most effective at removing indicator bacteria. Notably, indicator bacteria can survive for long periods in wetland sediments (Bavor et al. 2001), and be subsequently resuspended during wind or storm events to become a source of bacteria to surface waters (Kadlec and Knight 1996). For example, a study of the survival of indicator bacteria in wetland sediments showed that 24-27 days were required for 90% die-off of E. coli (Stenstrom and Carlander 2001).

There have been no studies to date that demonstrate that accumulation of indicator bacteria in wetlands or streams presents a risk to aquatic wildlife. Furthermore, relatively little is known about how wetland physicochemical parameters, hydrology, and biogeochemical processes affect the survival of indicator bacteria in sediments. Several studies have shown that under tropical conditions, such as those found in Hawaii or Guam, estuarine sediments can support the growth of Enterococci; however, there are no published reports of this occurring in semi-arid climates, such as southern California (Grant et al. 2001). Perhaps more of concern to the habitat value of wetlands for wildlife is increased antibiotic resistance from exposure trace contaminants in discharged wastewater and urban runoff, and potential exposure to wildlife pathogens. Increased resistance to antibiotics such as nalidixic acid, tetracycline, beta-lactam, and co-trimoxazole has been observed in the bacteria populations of urban streams (Goni-Urriza et al. 2000). While the major source of trace antibiotics is likely to be found in municipal wastewater, the heavy usage of antibiotics in the animal husbandry industry is likely to be of concern in treating nonpoint source runoff in wetlands with high populations of farm animals. Bacterial resistance to streptomycin and kanamycin were positively correlated with sediment mercury concentrations in some streams, suggesting an indirect selection for metal tolerance (McArthur and Tuckfield 2000). In general, very little is known about the concentrations of antibiotics in urban runoff, and the potential effects of these compounds on wetland biological processes.

Finally, recent research indicates that wetlands may act as a source of pathogens under certain circumstances. Grant et al. (2001) notes that Talbert Marsh, in Orange County California can be a source of bacterial loading to the near shore environment, and suggested that bird feces could be a source of pathogens to wetlands. Similarly, Gearheart (1999) found that passage of water from a constructed wetland to a wildlife enhancement area following UV disinfection resulted in an increase in indicator bacteria populations, with a seasonal signal correlated to the passage of migratory birds through the area. The impacts to humans and wildlife from these natural sources of pathogens is unknown. As natural and constructed wetlands are currently being considered as a treatment option to reduce high coliform counts in urban runoff of southern California watersheds, elimination of these indicator bacteria in wetlands sets up a potential conflict where provision of habitat for wildlife is also an objective. Clearly, improved understanding of the threats of animal sources of coliform bacteria to human health as well as threats of human sources of coliform to animal health are needed.

E. Summary

This chapter summarized the potential effects of dry season urban runoff and/or stormwater on the physical, chemical, or biological structure of habitat in natural or constructed treatment wetlands. Specific conclusions include:

- Increased urban runoff may result in changes in the magnitude, frequency, and timing of flow to streams and wetlands (e.g., increases in the magnitude and duration of peak flows, increases in the frequency of channel-forming events). Increases in urban runoff are generally proportional to increases in effective impervious surface (i.e., impervious surface that is hydraulically connected to a drainage system) and associated changes to the nature of the drainage network. This shift in hydrologic regime may result in destabilization of stream channels and shifts in plant community composition.

- Urban runoff can affect the physical structure of streams and wetlands by changing dry season flow patterns. The precise effect will vary depending on the geology of the watershed and the amount of impervious surface, as well as the amount of irrigation, vehicle and pavement washing, and POTW discharges. In some cases, decreased infiltration may reduce the period of the season in which baseflow is present. In other cases, dry season flow may increase due to increased impervious surface.

- The construction of wet ponds or dry extended detention basins tends to trap coarse sediments and interrupt the hydrological and sediment transport processes occurring on landscape scales. In particular, these features act as a barrier to the downstream movement of bedload, but still export fine-grained sediments. These effects can be especially detrimental when basins or ponds are created with a hydrological connection to the stream drainage network.

- Treatment wetlands may experience geomorphic alterations associated with changes in the amount or nature of sediment input to the stream/wetland. Changes in the nature or magnitude of sediment loads may result in aggradation or degradation of streams as they adjust to accommodate their new sediment loads, changes in particle size distribution, or changes in the topographic complexity of the substrate. The response of a specific stream to changes in the sediment regime will vary based on the nature and setting of the stream and the characteristics of the watershed.

- Increased runoff to natural or treatment wetlands may affect salinity. In freshwater systems, urban runoff my increase specific conductance, while brackish areas may become less saline due to the increasing magnitude and frequency of fresh water inputs

- Under *conservative* loading rates, many types of natural and constructed wetlands can sustainably assimilate nitrogen, phosphorus, and organic carbon inputs into the system. However, excessive loading may result in algal growth, large fluctuations in pH, chronically low dissolved oxygen conditions in surface waters, low redox potential (anoxic) sediments, and subsequent release of heavy metals, phosphorus, and organic contaminants previously sequestered in the sediments. High nutrient concentrations can also result in shifts in species composition of algal and plant communities toward species with higher nutrient tolerance.

- Accumulation of heavy and trace metals from urban wet and dry weather runoff is one of the greatest concerns in terms of potential bioaccumulation and ecotoxicity in treatment wetlands. Because many metals have an affinity for the surface area of particles, most of the metals in urban runoff are associated with suspended sediments rather than available in a freely dissolved form in surface waters. Studies have found that the highest levels of sediments are close to the inlets of the wetland s and that the management of sediments in these areas is the most critical.

- There is a strong correlation between the concentration of organic contaminants and suspended sediment, suggesting that sedimentation may be a major removal mechanism for organic contaminants in wetlands. Limited data exists on the sediment-water partitioning, transformation processes, and biological effects of organic contaminants in natural and constructed wetlands.

- Bacterial loading to wetlands and streams in urban areas has been well documented as one of the most common pollutants. Bacteria typically aggregate to the organic matter found on fine-grained suspended and bed sediments (< 2 µm fraction). Therefore, one of the main mechanisms for removal of bacteria from urban runoff in treatment wetlands is through the reduction of suspended sediment. Dieoff from UV exposure is also an important mechanisms for pathogen removal in wetlands..

- Recent research indicates that wetlands may act as source of pathogenic bacteria under certain circumstances. However, there has been little research on the risk to aquatic wildlife associated with accumulation of indicator bacteria in wetlands or streams.

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CHAPTER 4.
POTENTIAL EFFECTS OF URBAN RUNOFF ON WETLAND BIOTA

A. Introduction

The previous chapter summarized potential changes to the physical habitat of natural or created wetlands associated with urban dry weather and stormwater runoff. The next step in the ecological risk assessment framework is to evaluate risk to a particular species or community associated with exposure to urban runoff pollutants or other changes in the physical structure of the wetland (see Figure 2). In the case of treatment wetlands, key questions include:

- What is the potential that changes to a wetland's physical or chemical structure will affect the ecological health of the wetland in terms of its ability to support a diverse assemblage of typical wetland flora and fauna?
- How do potential changes in wetland structure impact the biological communities dependent on these wetlands, and are these impacts severe enough to be of concern? Do they outweigh the benefit of creating additional aquatic habitat with constructed wetland treatment systems?
- In the case of constructed wetlands, to what extent does the intentional manipulation of hydrologic regime, site morphology, and wetland biota to maximize water treatment capacity detract from the potential habitat value of the wetland?

In particular, concerns over the concentration of urban contaminants in wetlands and potential effects on wildlife have led some authors to conclude that "the use of wetland systems for treatment of contaminated surface water is generally incompatible with the goals of aquatic habitat enhancement" (Helfield and Diamond 1997). However, there are few documented effects of such contamination on wildlife frequenting treatment wetlands. Furthermore, attempts to assess the habitat value of wetland treatment systems through monitoring typically rely on documenting species and population densities of plants, macroinvertebrates, amphibians, reptiles, and birds as the measure of habitat value (e.g., USEPA 1993a). Species diversity and population counts are useful in evaluating the habitat function of treatment wetlands. However, there is a potential problem in interpreting these numbers as an indication of habitat quality when what is actually being measured is habitat attractiveness. 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).

The assessment of risk requires a more comprehensive approach in which the relationship between changes in wetland hydrology, site morphology, sediment dynamics, and water and sediment chemistry are related to ecological impacts on biota (Lemly 1997). We found no comprehensive assessments of risk to the biotic community of treatment wetlands in the literature, nor were any documented problems with treatment wetlands receiving urban runoff identified. Therefore, in this chapter, we frame the issue of risk by summarizing the existing literature that relates to potential impacts to wetland biota with respect to:

- Bioaccumulation and Toxicity
- Habitat and Species Diversity
- Community Structure and Food Web Dynamics

B. Bioaccumulation and Toxicity

Bioaccumulation of contaminants in the food chain and resulting toxicity to flora and fauna are the most cited risks in the discussion of potential dangers of utilizing wetlands to treat urban runoff (Helfield and Diamond 1997). However, little literature exists documenting the risks of nonpoint source pollution, such as urban runoff, on flora and fauna that utilize treatment wetlands. Furthermore, in most cases, contaminant residue in biota occur at levels below those linked to subacute effects in controlled toxicological studies; therefore, it is difficult to establish a direct cause and effect relationship between contaminant input and ecological health (Helfield and Diamond 1997). There is a particular paucity of literature pertaining to California climates, with the exception of some notable studies on selenium bioaccumulation and toxicity to aquatic life resulting from investigations of wildlife mortality in Kesterson Marsh in central California, and extensive literature on bioaccumulation in San Francisco Bay. Preliminary conclusions related to bioaccumulation and toxicity in wetland flora and fauna are summarized below.

Wetland Algae and Plants

Wetland algae and plants can extract and accumulate contaminants from surface waters and sediments, and in some cases also mediate the phytodegradation and transformations of these contaminants into either less or more bioavailable and/or toxic by-products (Susarla et al. 1999; Susarla et al. 2002). Certain algae and plants have been shown to hyperaccumulate metals and organic contaminants including nickel, zinc, copper, cadmium, selenium, benzene, ethyl benzene, toluene, xylenes, pentachlorophenol, short-chained aliphatic compounds and other contaminants (Susarla et al. 1999). Some plants can grow and tolerate hyperaccumulation while others may experience severe stress or die (Susarla et al. 1999). Kelly (1988) found enrichment ratios for metals in a variety of aquatic plants ranging from one to six orders of magnitude over background levels. Thus, where toxicity to the algae or plant does not occur, they have the ability to magnify concentrations, potentially making them available to wetland fauna at higher doses than measured in sediments or surface waters. While there is a sizeable database on bioaccumulation of contaminants by algae and macrophytes both in the field and in the laboratory (Aulio 1980; Hinman and Klaine 1992; Kelly 1988), little data is available for wetland and riparian species native to semi-arid and arid climates.

Historically, freshwater plants have been considered to be less sensitive than animal species to contaminants – leading to the preference of invertebrate and fish species for toxicity testing. However, studies that have compared the sensitivity of plants and animals have found the results to be chemical- and species-specific (Lewis 1995). Contaminants that were found to be more toxic to freshwater plants than animals include metals (e.g., Cr, Ni, Cd, Cu, and others), alcohols, pesticides (e.g., atrazine, aldrin, chlordane, and others), priority organics (e.g., phenols, chloramines, acrylates, and others), and many types of industrial effluents (Lewis 1995). However, studies of phytotoxicity of wetland macrophytes are relatively rare (Lewis 1995; Wang 1991), particularly in estuarine environments (Lytle and Lytle 1998; Lytle and Lytle 2001).

Freshwater microalgae and floating vascular plants such as duckweed *(Lemna* spp.) are the most commonly used to document phytotoxicity, and thus much of the available literature on responses of plants to nonpoint source pollution has been studied with these organisms (Davis et al. 2002; Lewis 1995; Van Dam et al. 1998; Vesk et al. 1999). Periphyton communities in freshwater wetlands and streams have been found to be extremely sensitive to phosphorus loading, with complete changes in community structure occurring in favor of more low-diversity, nutrient-tolerant assemblages (Hill et al. 2000; McCormick and O'Dell 1996). Genter and Lehman (2000) found in streams characterized by high concentrations of arsenic and copper that the diversity of the algal community was reduced with several species of diatoms and filamentous blue-green algae replaced by dominance by a single blue-green algal species.

While existing literature can be used to extrapolate the potential effects of urban runoff on treatment wetlands, risk will ultimately depend on loading rates, structure of the wetland, and tolerances of the specific species. A strong need exists for research documenting the existence of any such effects (and associated risk) on native wetland algal and plant communities in semi-arid climates such as southern California. More research is needed to understand the limits of tolerance in wetland flora to urban runoff, and how variation in wetland physical and biological processes affects these tolerance levels.

Wetland Fauna

General Findings

Bioaccumulation of contaminants and toxicological effects on wildlife have been documented for wetlands receiving nonpoint source runoff (Garcia-Hernandez et al. 2001b; Glenn et al. 1999; Ohlendorf et al. 1989; Schuler et al. 1990; Welsh and Maughan 1994). However, the presence and relative severity of effects depends on many site-specific factors, such as the wetland type, contaminant loading rates and concentrations, landscape position, hydrology, nature of sediment storage and transport, and structure of the biotic communities. There have been two fairly comprehensive reviews of bioaccumulation and toxicity in wetland fauna (Adamus et al. 2001; Wren et al. 1997). Most of the studies reviewed focused on effects on individual organisms rather than on biological communities; however, the results of these analyses help summarize several of areas of potential risk that should be investigated further at the community level. It should also be noted, that most studies were conducted in palustrine or lacustrine wetlands and relatively few studies have been conducted in riverine systems. A summary of the major findings is listed below:

- Organic contaminants and heavy metals such as mercury, lead, zinc, copper, and cadmium have been shown to be directly toxic to wetland invertebrates. They have also been shown to impact invertebrate communities by altering the species composition and abundance of algae and aquatic plants upon which invertebrates depend for food and shelter. 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. Exposure to contaminated sediments resulted in a slight reduction of survival of amphipods in controlled experiments, 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 noted that bioaccumulation and toxicity is a function of many site-specific factors, and that the mix of contaminants introduced, contaminant loading over time, and length of operation of stormwater ponds would be important in determining severity of effects found (Schueler 2000b).

- Metals in fish collected from Florida stormwater ponds were higher than in control ponds; however, there were significant differences between species. In general, bioaccumulation was greatest in benthic filter feeders, such as carp, and in carnivorous fish. Campbell (1995), in a study of several stormwater ponds in Central Florida, looked at trace metal concentrations in macroinvertebrates and fish species of three different guilds (bottom-feeding, predatory, and omnivorous). Bottom feeders that fed off of macroinvertebrates had the highest accumulation of metals, followed by predatory species, followed by omnivorous fish.

- Fish have been shown to bioaccumulate lipophilic species, such as PCBs and pesticides, but generally not at levels that pose a risk to consumer organisms (Adamus et al. 2001; Wren et al. 1997).

- Synthetic organics, including pesticides, have been shown to accumulate in wetland fish (Cooper 1991), often with adverse effects. In a Canadian wetland receiving oil sand effluent, fish had altered blood chemistry and died within 14 days (Bendell-Young et al. 2000). However, two herbicides used to control the invasive wetland plant, purple loosestrife (*Lythrum salicaria*), were not toxic to rainbow trout (Gardner and Grue 1996).

- Fish exposed to pesticides, PCB's, and other synthetic organics are often more vulnerable to disease. Dunier and Siwicki (1993) have presented a comprehensive review of this subject.

- Elevated levels of cadmium, lead, and mercury in stormwater ponds have been shown to interfere with reproduction of some amphibian species, most likely by affecting embryos and tadpoles (Adamus et al. 2001; Wren et al. 1997).

- Many synthetic organic compounds affect amphibians and aquatic reptiles. Petroleum derivatives have been noted to stunt tadpole growth of *Hyla cinerea* as well as reduce development time, growth, and survival in frogs and toads (Adamus et al. 2001; Wren et al. 1997)

- Studies on a suite of pesticides and insecticides show that adult amphibians seem generally unaffected by ambient concentrations; however, tadpoles and embryos are substantially more sensitive. Harris et al (1998) compared frogs in wetlands receiving runoff from an orchard treated with endosulfan to those outside of the orchard, and found no significant effects, despite accumulation of the substance in frog tissue.

- Several studies have documented high concentrations of organochlorines and PCBs in the tissues and eggs of several turtle species (Adamus et al. 2001; Wren et al. 1997). These species are relatively long-lived and relatively sedentary, and therefore, tend to accumulate contaminants over their lifetime.

- Waterfowl are attracted to wetlands and have the potential to accumulate contaminants, including pesticides, metals, and organic pollutants. There have been many studies on waterfowl contamination in wetlands and stormwater ponds. The specific results vary considerably depending on the particular pollutant and the species being studied. The most critical factor is assessing risk to waterfowl is length of exposure. For example, Wren (1997) reported that resident ducks feeding on contaminated marshes in Ontario had significantly higher concentrations of organic contaminants in the liver and muscle than migratory species.

- Similar to birds, mammals have been shown to accumulate metals and organic contaminants (Adamus et al. 2001). However, there are few studies documenting the effects of this accumulation in wild mammals

Examples in Arid and Semi-Arid Climates

Studies of Kesterson Marsh National Wildlife Refuge and San Francisco Bay estuary provide some of the most in-depth local examples of bioaccumulation and toxicity in wetlands. These cases should be viewed as worst-case scenarios, and cannot be extrapolated to all applications of the use of wetlands to treat nonpoint source effluent (e.g., from urban sources). This is especially true for riverine systems, which are flushed with much greater frequency and efficiency than the lacustrine or estuarine systems represented in these case studies. Nevertheless, these case studies are instructive in our understanding of the factors that can potentially lead to high rates of bioaccumulation and toxicity. These examples also stress the importance of undertaking a regional ecological risk assessment of possible biological impacts from utilizing wetlands to treat urban runoff to avoid potential adverse impacts and legal liabilities (Lemly and Ohlendorf 2002).

The 518-hectare Kesterson Marsh was created with the dual objective of treating agricultural irrigation drain water and providing habitat for wildlife (Zahm 1986). While the original project plan called for drainage water dilution of source waters through adjacent regulating reservoirs, depletion of project funds terminated construction of the master San Luis Drain, thus making the Kesterson Marsh the terminus of the irrigation wastewater (Zahm 1986). Surface water samples were saline and contaminated with selenium resulting from percolation through natural seleniferous soils of the San Joaquin River valley (15-400 $\mu g\ L^{-1}$). Selenium was found to heavily bioaccumulate in the food chain, resulting in 1,000-5,000 times the concentration of surface water in aquatic invertebrates and forage fish, and elevated concentrations in animal groups that utilized the marsh including fish, birds, insects, amphibians, snakes and small mammals (Clark 1987; Ohlendorf et al. 1988b; Ohlendorf et al. 1990; Saiki 1986). Toxicological effects included developmental abnormalities, embryo mortality, and reduced growth or survival of young, pathological changes in tissues, and chronic poisoning of wildlife (Lemly and Ohlendorf 2002). The Kesterson Marsh can be considered a worst-case scenario in treatment wetlands because of 1) high concentrations of selenium and other contaminants in effluent and extremely high contaminant loading rate, 2) arid climate with high evapotranspiration rate, 3) palustrine wetland with no outflow or dilution of effluent occurring, and 4) high concentration of wildlife. Similar characteristics and impacts can be found in the case of the Colorado River delta wetlands, where high rates of bioaccumulation and toxicity have also been observed in wildlife (Garcia-Hernandez et al. 2000; Garcia-Hernandez et al. 2001b; Welsh and Maughan 1994).

The tendency for estuarine wetlands to act as settling basins for watershed-derived contaminants also makes the wetland biota that utilize them vulnerable to bioaccumulation and toxic effects. Much research has been done documenting the extent of bioaccumulation heavy metals and organic contaminants in biota of San Francisco and San Diego Bays. In these estuaries, discharges from agricultural, industrial and urban nonpoint sources have resulted in heavily-contaminated sediments (Fairey et al. 1996; Long et al. 1988; Phillips and Spies 1988) and widespread bioaccumulation in both resident and migratory wildlife including macroinvertebrates including bivalves (Flegal 1977; Luoma et al. 1985), birds (Hothem et al. 1998; Hothem and Powell 2000; Lonzarich et al. 1992; Ohlendorf et al. 1988a), fish (Ohlendorf et al. 1985), and small mammals (Clark et al. 1992). Contamination of food sources is cited as reasons for poor body condition, reduced egg production, delayed ovulation, embryotoxicosis, and mortality of chick and adult resident and migratory birds utilizing these wetlands (Hothem et al. 1998; Hothem and Powell 2000; Lonzarich et al. 1992; Takekawa et al. 2002).

C. Changes in Species and Habitat Type Diversity, and Community and Trophic Structure

The use of natural and constructed wetlands to treat urban runoff can affect the biotic communities that utilize these wetlands by altering the diversity of species and habitat types, community structure, and trophic level dynamics. These consequences can arise either from 1) unintentional impacts associated with changes in the physical processes or structure of the wetland (e.g., hydrology, site morphology, sediment and water chemistry) due to increased urban runoff, or 2) intentional designs that are intended to maximize treatment capacity of a constructed wetland (e.g., manipulation of hydrology, wetland biota, and maintenance activities).

Effects on natural wetlands have been exacerbated by a combination of urbanization and increased regulatory requirements for mitigation measures to address degradation of water quality, increased urban runoff, and wetland loss. This situation has led to the restoration and creation of wetlands for the intended multiple roles of habitat replacement, water quality enhancement, and runoff attenuation. Each of these intended uses requires specific design features in terms of physical structure and plant community composition, which may or may not be compatible with the other intended uses. Because stormwater and sediment detention basins are among the most adaptable, effective, and widely applied BMPs in urbanizing areas (Schueler 2000a), they are often employed as 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 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 (Table 4). 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 Effects of this conversion on specific components of the wetland ecosystem are discussed in the sections below.

TABLE 2. Comparison of Features of Stream Habitat vs. Treatment Wetlands

Typical Features of Stream Habitat	Typical Features of Treatment Wetland
Short hydraulic residence time – flow through	Long hydraulic residence time – ponded
Woody, forested, or scrub-shrub	Emergent marsh and open water
Long, linear system/high edge to area ration	Large surface area/low edge to area ratio

Effects on Algae and Plants

Utilizing natural or constructed wetlands for treatment of urban runoff can affect algal and plant species diversity and community structure via changes in hydrology, sediment deposition, and surface water and sediment chemistry. The higher frequency and longer duration of flooding associated with urban stormwater treatment wetlands generally results in vegetation with lower species richness, with increased coverage of common, opportunistic (or invasive), and non-indigenous species (Azous and Cooke 2000; Thibodeau and Nickerson 1985; Van Der Valk 1991). With emergent and submerged aquatic species, an increase in drying events associated with large water level fluctuations can reduce species diversity (Leck 1996; Van Der Valk 1991) In addition, increased velocities associated with storm flows can uproot existing vegetation and hinder seed germination (Ewing 1996; Leck 1996). Ewing (1996) found that increased sediment deposition associated with urban runoff affected both sedges and sapling trees by reducing standing biomass and photosynthesis.

In cases where contaminants are phytotoxic to algae or plants, stress or death from toxicity can eliminate certain sensitive species, thus decreasing species diversity (Peterson et al. 1994). This is well documented in cases of the effects of increased nutrient loading on algae and plant communities in oligotrophic wetlands such as the Everglades, where increases in agricultural sources runoff of phosphorus have resulted in a complete community shifts in plants and periphyton (McCormick et al. 1996; McIvor et al. 1994). The effect of herbicides on aquatic primary producers such as algae and plants have also been well studied (Delorenzo et al. 2001; Maru 1993; Peterson et al. 1994). The elimination of key primary producers from a wetland can affect the higher-level consumers that are dependent on these algae or plants as a food source (Delorenzo et al. 1999; Delorenzo et al. 2001), potentially eliminating guilds or taxa from the food web.

In constructed wetlands, selection of the vascular plant palette is often based on the need to select species that are fast-growing, have a high tolerance for flooding and contaminant loading, and have a high uptake capacity for contaminants. Cattails (*Typha* spp.), reeds (*Phragmites australis*), bulrushes (*Scirpus* spp.), and rushes (*Juncus* spp.) for these reasons are favorites (Kadlec and Knight 1996; Schueler 1987). As a result, constructed treatment wetlands often have lower plant species diversity and a higher number of non-indigenous relative to native species than natural wetlands (Bonilla-Warford and Zedler 2002). In general, there is a lack of information on the suitability of native wetland plant species for use in treatment wetlands, a factor that plays into the reliance of engineers on the more common and predictable palettes of plants (Bonilla-Warford and Zedler 2002).

Use of streams and riparian areas to treat urban runoff may promote or hinder establishment of riparian habitat, depending on the specific circumstances. Numerous studies have investigated the relationship between hydrology and establishment of riparian species (Guilloy-Froget et al. 2002; Kozlowski 2002; Segelquist et al. 1993; Shafroth et al. 1998). The factors that are most responsible for successful establishment and growth of native riparian species are the timing of peak flows relative to the timing of seed dispersal, magnitude of peak flows in the year following seeding establishment, soil moisture characteristics, rate of water-table decline during seedling establishment, and maximum water table depth (Harry and Smith 1957; Scott et al. 1993; Stahan 1984). Appropriate flooding regimes bring desired nutrient-rich sediments to the floodplain, export organic and inorganic material from the floodplain, scour mature woodlands, and help spread propagules laterally into the floodplain, where they can establish on the fresh wet alluvial sand and gravel which is exposed following the flood season (Dickert and Tuttle 1985; Gosselink et al. 1990; Stahan 1984; Warner and Hendrix 1985). Alteration of flow patterns associated with increased urban runoff may promote establishment of opportunistic riparian species or invasive exotic plants (Everitt 1998). Since many riparian species are not extremely flood tolerant, their survival is dependent on stream conditions which will prevent them from being submerged for long durations (Scott et al. 1993), as may occur in situations where flow is augmented by urban runoff. Changes in storm flow patterns within a stream that is used for treatment may directly alter the distribution and diversity of riparian plant species by disrupting the complex spatial and temporal flooding and sediment delivery patterns upon which riparian species depend (Guilloy-Froget et al. 2002). Loss or riparian habitat diversity may also occur as a result of changes in channel geomorphology, such as incision or deposition (as previously discussed) that in turn preclude riparian plant establishment due to excessive scour or by raising floodplain elevations too far above the water table elevation (Kozlowski 2002).

Effects on Macroinvertebrates

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 use of wetlands for treatment of urban runoff has been shown to result in fairly well-documented effects on resident macroinvertebrate species diversity and community structure. 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 Chironimids, oligochaetes, and gastropods (Galli 1988; Ludwa and Richter 2000; Yousef et al. 1990).

A number of factors can contribute to changes in the composition of the macroinvertebrate community. Higher frequency and longer duration flooding and altered sediment transport dynamics (e.g., change in bed stability) favor species adapted to unstable habitats (e.g., Chironomids and oligochaetes, Collier 1995; Pedersen and Perkins 1986). Excess sedimentation in urban streams limits refugial space, so invertebrates are more susceptible to drift (Borchardt and Statzner 1990). Benthic organic matter, a major food source for stream invertebrates, has the documented capacity to bind many heavy metals and organic contaminants, thus exposing these fauna to potentially toxic effects (Benke and Wallace 1997). An increase in nutrient, organic matter, or contaminant concentrations in surface waters, sediments, or food sources has been shown to result in low diversity in macroinvertebrates, with an increase in the abundance of stress tolerant species. For example, chironomid and tubificid worms comprised over 90% of organisms surveyed in a central Florida stormwater pond (Yousef et al. 1990), while dipteran midge larvae constituted 95% of all organisms collected in muck layer of Maryland stormwater pond (Galli 1988). Despite the documented evidence of the effects of urbanization on wetland macroinvertebrate species diversity and community structure, Paul and Meyer (2001) note that little is understood about the effects of urban runoff on life cycle ecology, population dynamics, and community interactions of aquatic invertebrates in wetlands and riparian areas. Even less is understood about invertebrates native to wetlands in semi-arid and arid climates (Everts 1997).

Effects on Fish

While generally less is known about the impacts of urban runoff on fish species than on macroinvertebrates in wetlands, the literature generally presents similar trends with increased urban runoff: 1) decrease in fish species diversity and abundance, and 2) increase in the relative abundance of pollution tolerant taxa (Boet et al. 1999; Gafney et al. 2000; Onorato et al. 2000). Extirpation of fish species, particularly pollution-sensitive taxa, is common with increased urbanization in watersheds, as are increases in exotic species (Boet et al. 1999; Wolter et al. 2000).

The specific mechanisms responsible for fish community response to urban runoff in natural wetlands are less understood than for macroinvertebrates. Changes in hydrologic continuity or connectivity of streams or wetlands is often cited as a factor that may prevent fish migration or recolonization following impacts to up or downstream populations (Schueler 2000a). Placement of stormwater or sediment detention basins along stream reaches (e.g., in-line basins) can be present a substantial barrier to resident fish migration. Increased temperatures in larger ponds or wetlands with little vegetation or shading will favor establishment of warm-water fish communities, which in cooler climates are not native to local streams (Schueler 2000a). Changes in flow and sedimentation rates, particularly in streams, can affect fish habitat by removing or burying large logs and filling of pools and riffle structures used as spawning ground or refugia (Paul and Meyer 2001). Changes in salinity, dissolved oxygen concentration, algal biomass, and macroinvertebrate community structure associated with urban runoff have also been implicated in poor recruitment and large-scale fish kills, particularly in estuarine wetlands (Nordby and Zedler 1991). As observed for invertebrates, most data in the literature studying the effects of urban runoff on wetland fish species focuses on species diversity and relative abundance. There is little or no published information on behavioral ecology, community interactions, biomass, and reproduction of fish (particularly for non-salmonids) in urban streams and wetlands.

Effects on Amphibians

In their study of the effect of urbanization on amphibian populations associated with wetlands, Ritcher and Azous (2002) showed an inverse relationship between urbanization and species richness. Amphibians typically require particular types of substrates, submersed plants and flow patterns (e.g., shallow backwater pools) for successful deposition of their eggs and development and metamorphosis of juveniles (Adamus et al. 2001). Both prolonged desiccation and extreme floods can increase opportunities for invasion of streams or wetlands by exotic plant or animal species. Richter and Azous (2000) found that palustrine wetlands with water level fluctuations of 20 cm or less were significantly more likely to have a higher proportion of lentic-breeding amphibian richness. If patterns of vegetation become more homogeneous, as often occurs in treatment wetlands, the suitability of amphibian habitat as well as prey abundance may decline. Formation of deep, permanent pools may promote invasion by exotic predators, such as bullfrogs, which prey on native amphibian species. Changes in flow patterns to streams or wetlands can disrupt amphibian breeding by eliminating suitable breeding sites or forage locations. For example, the arroyo toad (*Bufo californicus*) requires shallow, slow-moving pools with fringing emergent vegetation for breeding and open sand bars for juvenile foraging (Sweet 1992). Changes in the depth or duration of high flows can hinder breeding and egg deposition for arroyo toad as well as for other amphibian species, and can redistribute woody debris and coarse sediments that are important components of amphibian habitat (Lind et al. 1996; Richter 1997).

Altered sediment regimes, which are typical of many treatment wetlands may also have an adverse effect on amphibian species. For example, deposition of silt, especially in combination with motor oil, resulted in reduced growth and earlier metamorphosis of larval tiger salamanders (*Ambystom tigrinum tigrinum*), as well as increased susceptibility of these species to *Saprolegnia* fungus (Lefcort et al. 1997). Studies of the effects of sedimentation in wetlands on amphibians reveal differing results. While many species require soft sediments as hibernation sites, excessive sedimentation and turbidity may impair light penetration through the water column and thereby inhibit growth of algae and submersed aquatic plants, which provide cover and attachment sites for amphibian eggs (Adamus et al. 2001).

Effects on Birds

Wetlands are an extremely important habitat for migratory and resident bird populations, particularly in semi-arid and arid climates (Page et al. 1997; Page et al. 1999). Urbanization alters the area of open water, existing hydroperiod, vegetation, water and sediment chemistry, and other wetland characteristics influencing cover, nesting habitat, and food quality and distribution of birds (Richter and Azous 2000; Weller 1994). While there have been few comprehensive studies looking specifically at the avian habitat value of natural or constructed wetlands used to treat urban runoff (Richter and Azous 2000), much of the literature of the effects of urbanization and nonpoint source contamination has relevance for this topic. In general, most of these studies find a shift in bird communities from predominantly native in undeveloped areas to a predominantly disturbance-tolerant generalist species in urbanized areas. Particularly notable are increases in predatory and aggressive species as well as exotic species in urbanizing landscapes (Gergel et al. 2002).

The published literature is most abundant documenting the effects of contaminated food sources on waterfowl. Predatory birds are the most susceptible to bioaccumulation of contaminants, with breeding failures associated with high levels of metals and organic contaminants (Farber et al. 1972). In Kesterson Marsh and San Francisco Bay, California, high concentrations of heavy metals (particularly selenium) and organic contaminants have been targeted as responsible for poor body condition, reduced egg production, delayed ovulation, embryotoxicosis, and mortality of chick and adult resident and migratory birds utilizing these wetlands (Hothem et al. 1998; Hothem and Powell 2000; Lonzarich et al. 1992; Takekawa et al. 2002). Other toxicological effects cited included developmental abnormalities, embryo mortality, and reduced growth or survival of young, pathological changes in tissues, and chronic poisoning of bird populations (Gordus 1999; Hoffman et al. 1988; Lemly 1996; Lemly and Ohlendorf 2002; Ohlendorf et al. 1990; Ohlendorf et al. 1989; Ohlendorf et al. 1988c). In many cases, it is difficult to link contaminant levels in body tissue and eggs to breeding success, because other factors can affect breeding success, including food availability, lack of appropriate breeding habitat and predation (Farber et al. 1972; Powell and Powell 1986). This is particularly relevant in urbanized environments, where many factors may be responsible for contaminant body burden and lower breeding success.

Changes in drainage patterns and some types of hydrological manipulation of streams and wetlands have been well-documented as contributing causes in the decline of many wetland bird species (Adamus et al. 2001; Colwell and Taft 2000; Johnsgrad 1956). Relative abundance of wading and shorebirds versus diving birds correlates significantly with wetland depth (Colwell and Taft 2000; Isola et al. 2000). Mortality of waterfowl eggs and young has been observed during nesting periods with strong fluctuations in water level (USEPA 1993b). In estuarine habitats, vegetation change associated with change in hydroperiod is associated with the lack of nesting habitat for the endangered Light-footed clapper rail (Zedler 1993). For this reason, the habitat value of stormwater treatment wetlands for certain species may be diminished as compared to natural wetlands because of widely fluctuating hydroperiods (Azous and Horner 2000).

Many studies have documented the influence of habitat structure, density, and complexity on avian nesting, foraging, and refugia. In addition, the width and composition of adjacent buffers has been recognized as influencing the suitability of a site for avian use (Bryce et al. 2002; Garcia-Hernandez et al. 2001a). For example, songbird community species richness has been shown to be higher in unfragmented riparian forest habitats than in mixed agricultural and urban sites, which typically have higher numbers of aggressive species, nest predators, and parasites (Richter and Azous 2000). Although the habitat requirements of specific species vary widely, the conversion of natural streams and riparian habitat with native buffers to stormwater treatment ponds in an urban matrix will generally decrease the biodiversity of native species inhabiting these sites.

Because birds often represent one of the highest trophic levels in streams or wetlands, they may integrate many of the other effects of urban runoff treatment previously discussed. For example, changes in the species composition and community structure of vegetation and macroinvertebrate communities may affect native bird populations. These effects can be subtle, slow to materialize, or (in the case of migratory birds) may not be apparent at the particular treatment wetland. Inappropriate habitat structure, hydro-regime, water depth, or substrate type may indicate that a treatment wetland is not providing appropriate food sources, nesting habitat, and refuge from predators (J. Skorupa, personal communication). Therefore, although they are attractive to birds, treatment wetlands might not necessarily be providing high quality habitat. Further studies of urban treatment wetlands are much needed to assess the impact of both obvious and subtle changes in habitat on native bird populations.

D. Summary

This chapter summarized how potential changes in wetland structure or processes associated with their use as treatment systems for urban runoff may impact the biological communities that are dependent on these wetlands. Specific conclusions include:

- Species diversity and population counts are an indication of habitat attractiveness but not necessarily of habitat quality. Such counts may not reveal subtle effects of urban runoff on habitat quality and biotic community health such as reproductive failure, weakened resilience from disease, inappropriate nesting habitat, refuge from predators, etc.

- Wetland algae and plants can extract and accumulate contaminants from surface waters and sediments, including metals and organic pollutants. Where toxicity to the algae or plant does not occur, they have the ability to magnify concentrations, potentially making them available to wetland fauna at higher doses than measured in sediments or surface waters.

- Heavy loadings of metals, pesticides, organic pollutants, and alcohols may result in toxicity to freshwater plants.

- Periphyton communities in freshwater wetlands and streams have been found to be extremely sensitive to phosphorus loading, with complete changes in community structure occurring in favor of more low-diversity, nutrient-tolerant assemblages.

- Typical urban pollutants, including metals, PCBs, and pesticides have been shown to accumulate in wetland macroinvertebrates, fish, amphibians, and birds. However, the severity of effects depends on many site-specific factors, such as the wetland type, contaminant loading rate, landscape position, hydrology, nature of sediment storage and transport, and structure of the biotic communities. In addition, previous studies have focused primarily on effects to individuals vs. communities, and few studies have been conducted in riverine wetlands (as opposed to palustrine or lacustrine wetlands)

- Studies of Kesterson Marsh National Wildlife Refuge and San Francisco Bay estuary provide some of the most in-depth local examples of bioaccumulation and toxicity in wetlands. The conditions that produced wildlife mortality at Kesterson should be viewed as a worst-case scenario (terminal wetland in a highly evaporative climate). However, it illustrates the importance of undertaking an ecological risk assessment of possible biological impacts from utilizing wetlands to treat urban runoff.

- The use of natural and constructed wetlands to treat urban runoff can affect the biotic communities that utilize these wetlands by resulting in a different diversity of species and habitat types, community structure, and trophic level dynamics.

- Because the physical and biological features that result in optimal stream habitat are very different that the optimal features for treatment of urban runoff, attempts to use treatment wetlands to fulfill the multiple objectives of habitat replacement, flood attenuation, and water quality control can result in "type-conversion" and overall degradation.

- Utilizing natural or constructed wetlands for treatment of urban runoff can affect algal and plant species diversity and community structure via changes in hydrology, sediment deposition, and surface water and sediment chemistry.

- Use of streams to treat dry season or stormwater runoff may result in changes in flow patterns and stream geomorphology (e.g., incision or aggradation). These changes may alter the distribution and diversity of riparian plant species by disrupting the complex spatial and temporal flooding patterns that they depend upon.

- The use of wetlands for treatment of urban runoff has been shown to affect macroinvertebrate species diversity and community structure by decreasing overall taxa richness, eliminating or reducing taxa belonging to scraper and shredder functional feeding groups relative to collector functional feeding groups, reducing the taxa richness and relative abundance of sensitive orders such as Ephemeroptera, Plecoptera, Odonata, and Trichoptera, and reducing or eliminating certain Chironomids taxa (with an increase in the abundance of tolerant Chironimids, oligochaetes, and gastropods).

- Increasing urban runoff to streams or wetlands may result in a decrease in fish species diversity and abundance, and an increase in the relative abundance of pollution tolerant taxa. This is due mainly to changes in hydrologic continuity or connectivity of streams or wetlands, increased temperature, and changes in flow or sedimentation rates.

- Changes in flow and sedimentation patterns to streams or wetlands associated with their use a treatment systems can disrupt amphibian breeding by eliminating suitable breeding sites or forage locations.

- Because birds often represent one of the highest trophic levels in streams or wetlands, they may integrate many of the other effects of using streams or wetlands to treat urban runoff. Habitat suitability for birds may be affected by contaminant loading, changes in hydrologic regime, changes in plant community composition and structure, and loss of native riparian or upland buffers.

The replacement of natural wetlands and associated riparian and floodplain habitat with stormwater treatment wetlands and detention ponds results in a reduction of the diversity of habitat types found on the landscape as well as within the wetland itself. Flora and fauna endemic to southern California wetlands have evolved life cycles that are dependent on the complex microtopography and mosaic of wetland and upland habitat types in the landscape (Ferren et al. 1996). The loss of habitat diversity results in an overall reduction in native species diversity, and a loss of habitat value for the region as a whole (Sutula et al. 2002). On the other hand, there is an urgent need for the implementation of urban runoff BMPs (treatment wetlands are not the only BMPs in the toolbox), because storm flows are primary agent in the deterioration of wetlands located downstream of urban areas as well (Schueler 2000a). A balance must be achieved through careful formulation of policy and guidelines to guide the recovery of natural wetlands and riparian areas and the implementation of urban runoff BMPs in the watershed (Schueler and Holland 2000b).

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CHAPTER 5.
SITING, DESIGN, AND MANAGEMENT OF TREATMENT WETLANDS: RECOMMENDATIONS FROM THE LITERATURE

A. Introduction

In southern California, many natural, restored, or created wetlands are at least partially supported and sometimes caused by urban runoff, even if their primary purpose is not for treatment. Furthermore, many degraded and fragmented wetlands no longer have a reliable water source due to upstream water allocations or depleted groundwater tables. Pretreated effluent from wastewater treatment plants, seasonal return irrigation flows and other types of urban runoff may be the only sources of water available for many aquatic ecosystems in urbanizing watershed. Notably, many have called for considering stormwater a resource for managing wetland habitats (NTS Master Plan 2003). Encouraging the construction of structural BMPs including treatment wetlands will likely substantially increase the quality of the habitat of downstream wetlands. Given this reality, it is important to develop wetland design and best management practices aimed at mitigating potential risk to wildlife that utilize urban or treatment wetlands. These guidelines should be part of broader regional policy that addresses the need for comprehensive landscape planning, stormwater management, and wetland protection. An excellent starting point for regional guidelines are those published by Horner et al. (2000). These guidelines include:

- Comprehensive landscape planning for wetlands and stormwater management including 1) comprehensive planning steps and 2) stormwater wetland assessment critieria
- Wetland protection guidelines including: 1) general wetland protection guidelines and 2) guidelines for protection from adverse impacts of modified runoff quantity discharged to wetlands.

The below recommendations should be not be considered a complete or detailed compilation but rather a starting point for future discussion with stakeholders. Any guidelines adopted must be constantly revisited as additional research is completed.

B. Landscape Planning Considerations

The US EPA strongly recommends that the siting of treatment wetlands be part of a watershed-wide planning effort that includes a water quality management strategy (USEPA 2000a). Potential on-site as well as cumulative impacts and benefits should be carefully considered. Specific recommendations found the literature include:

- Maximize water storage and infiltration opportunities within the landscape unit and outside of existing wetlands to minimize urban runoff (Horner et al., 2000)

- Preferably install several smaller decentralized treatment wetlands throughout the watershed rather than one large treatment wetland (Schueler and Holland 2000b).

- 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 hydrologically-isolated from surface water flow rather than within the historic floodplain (Schueler 2000a).

- Given evidence that the presence of riparian buffer strips greatly reduce contaminant concentrations in runoff before it is discharged to streams and wetlands (Schueler and Holland 2000b), the installation of treatment wetlands in a watershed should be coupled with an aggressive program to restore riparian buffers around streams and wetlands (Schueler 2000a). Also maintain interconnections between wetlands and other natural habitats (Horner et al. 2000).

- 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).

- Attempt to preserve mature riparian forest or habitat area. Limit area of disturbance and mandate tree protection measures during construction (Schueler 2000a).

- 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).

- Consider alternatives 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).

- 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 (H. Ohlendorf, personal communication). 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.).

C. Design Considerations

A well-conceived design for a constructed wetland is critical not only to mitigate any potential impacts to wildlife but also to assure a high level of treatment performance. As noted in Chapter 2, there is a great need for additional research and guidance to be developed for the design of urban runoff wetlands design and operations manual for semi-arid and arid environments. The design considerations given below should be considered a starting point for discussions with stormwater engineers, agency staff and stormwater management agencies to further develop design and operating guidelines for southern California.

Hydraulic and Contaminant Loading Rates

Currently, the bulk of performance data guiding the design criteria for treatment wetlands and stormwater ponds is from humid climates. Research is needed to improve the design criteria specific to semi-arid environments. Given this lack of guidelines appropriate for regions such as southern California, it is important that treatment wetlands be sized to allow for conservative hydraulic and contaminant loading rates (Caraco 2000).

Importance of Source Control and Pretreatment

Treatment wetlands should not be regarded as the "silver bullet" to resolve all water quality problems in a watershed. There is much evidence that suggests that for stormwater management, treatment wetlands can only be effective long-term if incorporated with a strategy of aggressive source control and pretreatment to reduce contaminant concentrations (Caraco 2000). High contaminant loading rates can harm wildlife, and result in deterioration of wetland function and loss of treatment capacity (USEPA 2000a). Literature suggests that wetland should not be used as the first interceptor of urban runoff, particularly in industrial or highly urbanized sites (Schueler 2000a). Specific recommendations include:

- Control 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, provide pretreatment including:

    - Oil and grit interceptors in highly industrial sites, highways, etc. (Shutes et al. 1997),

    - Place a primary treatment system such as a sand filter or forebay prior to wetland and install floating berms to trap trash and large debris.

    - Utilize sediment collection/settling forebays for treatment of stormwater inflows and for additional treatment of wastewater. Design and locate the forebays for ease of maintenance, removal of sediment, and to achieve greatest protection of wetland habitat and receiving waters (Schueler 2000a).

Segment the wetland 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.

- Incorporate a variety of treatment strategies, including detention, pre-treatment, treatment, and infiltration, in series to maximize removal efficiencies and minimize exposure to wildlife.

Maximize Native Habitat and Plant Diversity

Developing a wide variety of habitat type within a treatment wetland will enhance higher wildlife diversity. Controls in the design of the treatment wetland that affect habitat type and plant diversity include seasonal hydroperiods, depth-flow changes, vegetative succession, and accumulation of sediments (USEPA 2000a). Habitat quality can be enhanced by providing for areas of open water, creating nesting islands for waterfowl, and leaving some upland and buffer areas for other nesting species. Because the selection of plant palette is one of the major controls on habitat quality in a treatment wetland, it is important that the following recommendations be considered in the design:

- Where appropriate, design constructed treatment wetland to provide habitat with a diversity of native species comparable to similar wetlands in the region (Bonilla-Warford and Zedler 2002; USEPA 2000a).

- Create gentle slope to allow for good plant establishment and diversity. Design for moderate water level fluctuations (McLean 2000).

- Maximize vegetative species diversity, where appropriate, without increasing the proportion of weedy, non-indigenous, or invasive species at the expense of native species. Project plans should include mechanisms to control or eliminate undesirable species. Additional research is needed to determine which native upland and wetland plant species are compatible for treatment wetland use.

D. Maintenance

Long-term, regular maintenance of treatment wetlands is critical to sustain treatment capacity and optimize the habitat value provided (Kadlec and Knight 1996) and should be required indefinitely. Maintenance can include cleaning of pretreatment areas (dredging of sediment forebays, trash removal, backwashing of sand filters, etc.), harvesting of plant biomass, removal of exotic species and replanting of desired species. All maintenance work must be scheduled to avoid critical breeding and nesting periods for wetlands species.

E. Importance of Monitoring and Adaptive management

US EPA strongly recommends the need for monitoring of the treatment wetlands to ensure that the system is functioning properly and not becoming an attractive nuisance to wildlife (USEPA 2000a). This monitoring data should be used to adaptively manage the treatment wetland, potentially identifying problems, improving performance, and enhancing habitat value. While almost all pond or wetland monitoring studies have been one-time or limited snapshots taken over a few years at most (usually right after construction), efficiency of treatment can change dramatically with time, thus arguing the need for long-term or at least period monitoring on older systems. Investigations of treatment efficiency in stormwater ponds 9-10 years following construction found that wetland species had changed dramatically with original species supplanted by invasive species and a decline in nutrient and sediment removal capacity (Oberts and Osgood 1991).

Currently, there are no established criteria for monitoring of wetlands in southern California, including constructed or treatment wetlands (Sutula et al. 2002). Wren et al. (1997) provide a detailed summary of suggested monitoring requirements for constructed wetlands, presented as a sequential series of monitoring steps organized around the risk assessment paradigm (see Chapter 3). Each step leads to more detailed investigation, if ecological impacts are shown or expected. Their recommended monitoring includes baseline monitoring and Level I and II monitoring associated with risk characterization. Baseline monitoring is ongoing, while Level I is conducted periodically to assess the potential risk and determine the need for more intensive monitoring. Recommended baseline monitoring includes:

- Wetland size (surface area)
- Water depth
- Inflow volume
- Expected retention time
- Inflow/outflow water quality (wet season and low flow)
- Sediment contamination
- Toxicity testing

Level I monitoring includes:

- Vegetation and wildlife habitat evaluation
- Receptor identification (i.e. identify benthic community, wildlife use)
- Contaminant analysis of benthos and fish (if warranted)

Level II monitoring is conducted if determined necessary by baseline and Level I data, and would normally include intensive studies focusing on a clearly identified potential species or groups at risk.

The guidelines developed by Wren et al. (1997) can serve as an excellent starting point for discussion to create minimum monitoring requirements for treatment wetlands in southern California. This monitoring should be consistent and compatible with the National Stormwater Best Management Practice Database (www.bmpdatabase.org) to ensure that all data generated can be used for evaluation of BMP effectiveness in the future. In addition, all projects should be reviewed by a central person or committee that oversees the research program to determine if they can support the research program.

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CHAPTER 6.
SUMMARY AND RESEARCH RECOMMENDATIONS

A. Summary of Findings

Urban development of large portions of coastal southern California has resulted in increased runoff being generated during both the wet and dry seasons. The increased runoff to natural wetland and riparian areas may have both positive and negative impacts on the physical and biological structure of the ecosystem. While increased runoff to streams may result in scour and associated loss of wetland habitat, urban runoff may be the only reliable source of water to support restoration of habitat where groundwater tables have been depleted. Furthermore, the necessity of improved water conservation and surface water quality in coastal waters has created a demand for strategies to retain and/or treat the runoff prior to its discharge to a receiving waterbody. Over the last 10-15 years treatment strategies and BMPs have increasingly incorporated the use of natural or constructed wetlands. Many scientists and policy-makers have argued that, given the reality of anthropogenic water resource use in arid and semi-arid climates, restored wetlands or constructed treatment wetlands whose principal water source is urban runoff may provide tangible ecological (and water quality) benefits in regions with high loss of aquatic habitats (USEPA 2000a). Certainly, the construction of treatment wetland in upland areas increases the amount of wetlands habitat overall, and could arguably increase the habitat value of natural wetlands downstream. Ultimately, the determination of the net benefit or detriment of natural or created wetlands that rely on an altered hydrologic regime will depend on the specific nature of the system and the assessment endpoint.

To better understand the complex issues associated with the habitat quality or hazards associated with natural or constructed wetlands subject to urban runoff, we posed the following questions:

1. Do wetlands created or enhanced for water quality purposes become environmental hazards for wildlife? If so, under what circumstances?

2. How does directing urban runoff or stormwater through or a natural or restored wetland affect the habitat of the wetland?

    a. What is the potential that changes to a wetland's physical or chemical structure will affect (positively or negatively) the ecological health of the wetland, in terms of its ability to support a diverse assemblage of typical wetland flora and fauna?

    b. How do potential changes in wetland structure impact the biological communities dependent on these wetlands, and are any negative impacts severe enough to be of concern?

3. How do considerations of the habitat effects of treatment wetlands differ by wetland class (i.e., how are the issues different for riverine vs. palustrine or lacustrine wetlands)?

Unfortunately, there is little direct research on these questions. Consequently, we have drawn on a variety of related literature to extrapolate general conclusions and to help provide insight into potential effects and key considerations. This chapter summarizes the areas in which future research is needed.

Review of over 200 reports, articles, books, and other references allowed us to draw the following general conclusions:

- *Adequate research does not exist to address the three central questions given above.*
  The research literature generally falls into two camps that are reflective of wastewater engineering and natural science disciplines respectively: 1) What is the efficacy of treatment wetlands in removing contaminants and how can this removal be maximized? 2) What are the effects of urbanization on wetland physical attributes, flora and fauna? Neither body of literature adequately addresses the above questions.

- *Documented examples exist of potential incompatibility between water quality and habitat goals in wetlands (e.g., Kesterson Marsh). Overall, the literature reviewed suggests concerns of risk to wildlife are valid, but because these studies were not specifically designed to address this question, the degree of risk is unknown.*

  Research on the effects of urbanization on wetlands has found that urban runoff can affect the biotic communities that utilize these wetlands by altering the hydrology, biogeochemistry, diversity of species and habitat types, community structure, and trophic level dynamics. It is not clear however, given the extreme loss of aquatic habitat in southern California, that the risks posed by such a use are outweighed by the additional benefit provided by the restoration or especially the creation of aquatic habitat (albeit for water quality purposes). Ultimately, the determination of the net benefit or detriment of natural or created wetlands in treating urban runoff depends on a host of site-specific factors including contaminant and hydrologic loading rates, upstream BMPs, wetland type, geology and land use in the watershed, landscape position, hydrology, nature of sediment storage and transport, and structure of the biotic communities. Although the literature reviewed in this report is useful in considering how contaminants from nonpoint sources may affect biological communities, the question of how severe this impact may be in treatment (natural or constructed) wetlands in southern California cannot be directly extrapolated from this information.

- *There is a general lack of literature on effects of urbanization on wetlands or efficacy of using wetlands to treat urban runoff in semi-arid/climates such as southern California.*

  Most of the literature that addresses efficacy of treatment or effects of urbanization on wetlands has been conducted in humid, temperate climates. This literature would be helpful in addressing the risk posed to wildlife from treatment wetlands, but is less applicable to local conditions given southern California's semi-arid climate. This argues for additional research specific to local conditions. In addition, it is recommended that a database of local treatment wetland projects be established in order to evaluate existing monitoring data and establish potential sites for future research.

B. Future Research Needs

A risk assessment framework is a useful paradigm to examine the potential hazards associated with the presence of contaminants in constructed wetlands or changes in wetland structure associated with increased runoff and loading. In this section, future research needs are organized by risk assessment component: 1) exposure assessment, in which the level of contamination or damage to the habitat is quantified, and 2) ecological effects assessment, in which the effect of the damage or contamination to a particular species or community is assessed. An additional research theme is included addressing knowledge gaps related to mitigating potential risk to wildlife from treatment wetlands. Given that different government agencies or organizations may have different priority research needs, no attempt was made to prioritize the list of research needs. Rather, prioritization of a future research agenda should be done via a consensus process involving all appropriate entities.

Exposure Assessment

This set of research questions is intended to improve our understanding of the effect of urban runoff and the associated pollutant loading on the physical and chemical structure of wetlands in Southern California. Understanding the exposure and loading associated with urban runoff is critical to determining the assimilative capacity of wetlands to pollutants and their resiliency to changes in hydrology. This in turn will help decision-makers determine under what circumstances urban runoff can be directed to wetlands without denigrating their physical or chemical structure. Specific research questions are:

5. What are typical wet and dry season loading rates and concentrations of various pollutants from specific land use types? The design of a research questions to address this questions should review existing information currently collected by SCCWRP, and the Los Angeles, Ventura, and Orange County stormwater monitoring programs.

6. What is the water quality enhancement capability of treatment wetlands (constructed and natural) and riparian areas in semi-arid/arid climates under a range of contaminant concentrations and loading rates? How do the effluent

qualities from these wetlands change over time (1 year, 3 years, 5 years, 10 years, 20 years)?

7. What are the natural background loading rates and concentrations of various constituents (nutrients, heavy metals, etc.) in the surface waters of wetlands and riparian areas in southern California? How do they vary spatially (both by position within the watershed and between watersheds) and temporally over seasonal or climatic cycles? How are these questions different for constructed versus natural wetlands?

8. How does diversion of dry weather flow and/or stormwater into wetlands and riparian areas alter (or establish in a new constructed wetland):
   a. physiochemical characteristics (pH, conductivity, alkalinity)?
   b. sediment deposition rates, texture and grain size distribution, and organic matter content?
   c. hydrologic regime including seasonal and annual water budgets and surface water and groundwater levels?
   d. Concentrations and seasonal/annual budgets of contaminants?

5. For each wetland class, what is the maximum allowable change in peak flow rate and duration, water level fluctuations, average flow volume, and other changes in hydrologic regime that can result without deleterious effects to the wetland habitat values?

Ecological Effects Assessment

This set of research questions is intended to improve our understanding of potential biological effects of increased runoff or loading, in terms of wetland flora, fauna, and community structure. Understanding potential hazards will better define tolerances of target species and biotic communities and conditions that are either conducive or deleterious to faunal use of treatment wetlands. This will help decision-makers assess the desirability of using urban runoff to help support wetland habitat and the potential risks to wildlife associated with these practices. Specific research questions are:

7. What are the most appropriate assessment endpoints to evaluate risk (e.g., plant chlorosis, decreases of microbial communities, changes in species composition, reproductive failure)?

8. How does the plant community composition and trophic structure of treatment wetlands compare to natural wetlands?

9. What is the habitat value of wetlands and riparian areas restored/created to maximize water quality benefits, in terms of species structure, abundance, and diversity?

10. What is the sensitivity of key plant or animal species to contaminant loading and accumulation? Sensitivity must be defined by toxicological endpoints such as acute and chronic toxicity, breeding, nesting, or reproductive success, etc.

11. How are plant communities affected by changes in hydrology or sediment delivery associated with urban runoff?

12. What are the rates of bioaccumulation of contaminants at various trophic levels?

13. Is toxicity observed in macroinvertebrates, fish, amphibians, or birds that live, forage, or breed in wetlands exposed to urban runoff? Are specific life stages or species more sensitive than others? What are the most appropriate biological indicators for monitoring toxicity?

14. What is the habitat value of treatment wetlands on a landscape scale?

9. How do various treatment wetland design attributes reduce potential hazards and increase habitat values in various components of treatment wetlands?

Research on Mitigating Risks to Wildlife and Maximizing Habitat Benefits in Treatment Wetlands

This set of research questions is intended to improve our understanding of how specific design or maintenance practices affect the overall risk to wildlife associated with treatment wetlands. This information will help improve our ability to minimize risk and optimize wetland management for both wildlife and water quality goals.

7. What are the appropriate design criteria for wetlands treating wet weather and dry weather urban runoff if providing habitat is a primary or secondary objective?

8. What are the appropriate design criteria if the objective is to construct a wetland for treatment and not attract species that could be harmed?

9. What other treatment wetland design attributes (length to width ratio, etc.) contribute to improving habitat in wetlands?

10. What is the effect of pretreatment (sediment forebays, sand filters, etc.) on reducing contaminant loading to the main wetland or riparian area? What is the extra burden in cost and labor that such pretreatment strategies impose?

11. To what extent does routine maintenance in forebays and main wetland cause a disturbance or diminish habitat value? What are recommended ways to minimize this disturbance?

12. What plant species native to southern California are most suitable for cultivation in treatment wetlands (i.e., the most flood and contaminant tolerant)? What are

the recommended native plant palettes with respect to different habitat types within a treatment wetland (shallow water versus deep water, etc.)?

13. What source control or other treatment BMPs (upstream of wetland) are appropriate as part of an overall water quality program?

Special Considerations in Designing Research Program

Considerations for the design of a research program should include the establishment of a database of local treatment wetland projects in order to evaluate existing monitoring data and establish potential sites for future research. Furthermore, the design of future research projects on habitat value of treatment wetlands in southern California should be comprehensive and should include the following:

- A broad suite of constituents should be evaluated, including pH, conductivity, temperature, sediment grain size distribution, heavy metals, nutrients, priority organics, pathogenic bacteria, organic matter content, and algal biomass.

- Concentrations should be evaluated in various "compartments" of the wetland ecosystem (e.g., sediment, water column, plant tissue, primary consumers).

- Patterns and effects should be evaluated in multiple wetland classes (e.g., palustrine, riverine), on multiple types of treatment wetlands (e.g., wetlands used to treat dry season runoff and/or stormwater), and under a range of different contaminant concentrations and loading rates.

- Research should address effects on the structure of biologic communities as well as on individual species. Furthermore, because the potential risks to wildlife may be subtle, it is important to assess impacts at various spatial scales including that of the site, watershed, and ecosystem (Leuven and Paudevigne 2002). These studies should not be limited to population abundance but should also include an evaluation of more subtle effects such as effect on reproductive success, nesting habitat, etc.

- Ecotoxicological effects in aquatic dependent species from multiple trophic levels (e.g., macroinvertebrates, amphibians, birds, mammals) should be addressed.

- Pattern and effects should be evaluated over time, especially following episodic events, such as heavy rainfall.

C. <u>Conclusions</u>

Many wetlands and riparian areas in Southern California receive urban runoff and therefore function either intentionally or inadvertently as both habitat and water quality treatment areas. As development continues to expand to new areas and urban centers are redeveloped there is likely to be increased interest in the ability of wetlands to perform this dual role. Therefore, it is critical to initiate a coordinated research program to better understand the tradeoffs between habitat and water quality functions and improve our ability to design and manage treatment wetlands appropriately. In the interim, better monitoring of the physical, chemical and biological attributes of treatment wetlands compared to natural wetlands will improve our understanding of the benefits and risks associated with wildlife use of these systems.

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