WETLANDS & WATERSHEDS

Direct and Indirect Impacts of Urbanization on Wetland Quality

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Direct and Indirect Impacts of Urbanization on Wetland Quality

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Article 1 of the Wetlands & Watersheds Article Series

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Executive Summary

This article synthesizes more than 100 scientific studies on the direct and indirect impacts of urbanization on wetlands and the key role wetlands play in watershed quality. Some of the major findings include:

- Wetlands provide important ecological services that contribute to watershed functions, most notably in pollutant removal, flood attenuation, groundwater recharge and discharge, shoreline protection, and wildlife habitat. The benefit of wetland ecological services generally increases as total wetland cover increases in a watershed. Numerous researchers have quantified the economic benefits provided by wetlands in a watershed. When wetlands are lost or degraded by land development, these services must often be replaced by costly water treatment and flood control infrastructure. Given the many watershed services wetlands provide, wetland conservation and restoration should be an integral part of a comprehensive local watershed management strategy.

- Although the national rate of wetland loss has dropped sharply in recent years, the goal of no net loss in wetland quality remains elusive. Development in urban and rural areas now is the cause of more than 60% of national wetland loss. Several national assessments have noted deficiencies in current federal and state regulatory programs that allow direct and indirect impacts to wetlands that reduce their function and quality to continue. These regulatory gaps can best be closed by increased local management and regulation of wetlands.

- More than 50 of the studies reviewed document indirect impacts to wetlands caused by land alteration in the contributing drainage area to wetlands. Upland development increases stormwater to wetlands, and downstream crossings create flow constrictions. Together these changes lead to increased ponding, greater water level fluctuation and/or hydrologic drought in urban wetlands. In addition, urban wetlands receive greater inputs of sediment, nutrients, chlorides, and other pollutants; concentrations in urban stormwater are typically one to two orders of magnitude greater than predevelopment conditions (Schueler, 1987).

- Numerous studies describe how urban wetlands respond to these stressors. Although the precise response depends on the sensitivity and landscape position of the wetland, the

general trend is a sharp decline in the diversity of the native plant and animal community and an increase in invasive plant species that can tolerate stressed conditions. Research has shown that degraded urban wetlands lose many of their important watershed functions. The indirect impact of upland development on wetlands is currently not regulated by state or federal agencies.

More research is needed to fully define the indirect impacts of land development on wetlands, and several priority research strategies are outlined at the end of the article. Taken as a whole, however, the current science on wetland impacts from development presents a strong and persuasive case to support greater local regulation and management of wetlands and their contributing drainage areas. The recommended local watershed planning approach to wetland management is outlined in Articles 2, 3 and 4 of this series.

About the Wetlands & Watersheds Article Series

The Wetlands & Watersheds article series was developed by the Center for Watershed Protection (CWP) in cooperation with the United States Environmental Protection Agency (USEPA). Funding for this project was provided by USEPA under cooperative agreements number CD-83192901-0 and WD-83264101-0.

Collectively, wetlands provide many watershed benefits, including pollutant removal, flood storage, wildlife habitat, groundwater recharge, and erosion control. While watersheds and wetlands are interconnected systems, their management is often segregated along regulatory and jurisdictional lines. Recent initiatives, such as the National Wetlands Mitigation Action Plan, provide a potential framework to integrate wetland protection in the context of larger local and state watershed planning efforts. However, no specific guidance exists for managing wetlands in the context of local watershed plans, and local governments often lack the tools and knowledge to effectively protect critical wetlands. This project was designed to fill this gap by expanding CWP's current watershed protection guidance, tools, and resources to integrate wetlands into larger watershed protection efforts. A key message conveyed in this new guidance is that wetlands should not be managed separately from other water resources because they are integral to water resource management.

This project included *research* on urban wetlands and local protection tools, *synthesis* of the research into a series of articles, and *transfer* of wetland protection tools and resources to wetland and watershed professionals across the country. The audience for the articles includes local natural resources managers and land planners who would benefit from guidance on local tools for protecting wetlands. The Wetlands & Watersheds article series includes the following:

Article 1: Direct and Indirect Impacts of Urbanization on Wetland Quality
This article reviews the direct and indirect impacts of urbanization on wetlands, and describes how impacts to wetlands affect watershed health.

Article 2: Using Local Watershed Plans to Protect Wetlands
This article presents detailed methods for integrating wetland management into the local watershed planning process.

Article 3: Adapting Watershed Tools to Protect Wetlands
This article describes 37 techniques for protecting wetlands through local programs and ordinances.

Article 4: A Local Ordinance to Protect Wetland Functions
This article outlines the key elements of an effective ordinance to protect existing wetlands from the indirect impacts of land development by regulating land use in their watersheds, and provides adaptable model ordinance language.

Article 5: Urban Wetland Restoration Techniques
This article features a watershed approach to identify and assess priority sites for wetland restoration and creation in urban areas.

Article 6: The Importance of Protecting Vulnerable Streams and Wetlands at the Local Level
This article makes the case for expanded local protection of vulnerable wetlands and streams that may not be fully protected by state or federal law due to their small size or geographic isolation.

Other wetland-related products of this project include wetland slideshows, an annotated bibliography of wetland research, a listing of key wetland web resources, and more products available on the newly expanded wetlands section of the CWP website at http://www.cwp.org

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Introduction

The purpose of this article is to review the current state of wetland science as it pertains to impacts from urbanization, and to explore the possible management implications for local natural resource managers and land use planners who are the principal audience for this article. It is always difficult to generalize about wetlands because they are so diverse with respect to their hydrology, plant communities and landscape position. In a real sense, no two wetlands are similar in their quality or function. In addition, the nature of urbanization (also referred to as land development in this article) in the area draining to wetlands often differs greatly from site to site. Despite this variability, several consistent and recurring impacts have been observed in different wetlands around the country. At the outset, it is important to define the terminology that will be used throughout this article. Box 1 provides definitions for various terms used in this article and subsequent articles.

The scope of this literature review includes more than 100 research studies and reports on the relationship between land development and wetland quality. This topic is certainly an emerging area of interest, as more than 40% of the research was published in the last five years. The research was scattered among a very wide range of academic disciplines that do not generally interact with each other – hydrology, herpetology, landscape ecology, botany, wildlife management, conservation biology, toxicology, stormwater management and wetland science to name a few. In addition, researchers have worked in many different regions of the country, examined many different wetland types and utilized a wide range of assessment methods. Consequently, this is the first national review to synthesize this diverse and scattered literature into a comprehensive and unified framework.

This article synthesizes more than 100 scientific studies on the direct and indirect impacts of land development on wetlands and the key role wetlands play in watershed quality. First, it describes some of the functions and services wetlands provide and summarizes available replacement cost data when these services are lost. Next, the article discusses how land alterations introduce specific stressors to wetlands, such as increased sediment and nutrient loads, changes in hydrology, and disconnection with the floodplain. These stressors ultimately affect the function and condition of the wetland, so the rest of the article illustrates the link between these stressors and the ultimate impacts they have on wetland plant, animal and vegetative communities.

Box 1. Important Wetland Definitions Used in this Article

- *Contributing Drainage Area (CDA)*: Specific landscape features that transmit water to wetlands (e.g., surface drainage areas, groundwater recharge areas, stormwater outfalls)

- *Impact*: Response of a wetland to one or more stressors. Examples include changes in water level fluctuations, die-out of native species, decline in amphibian abundance and diversity, and growth of invasive plants.

    - Direct Impact*:* wetland loss or degradation resulting from activities that occur within wetlands, such as dredging, filling and draining. Activities that cause direct impacts are largely regulated through the federal and state wetland permitting process.
    - Indirect Impact: Impact to wetlands caused by inputs of stormwater and pollutants generated by land development or other activities within the wetland CDA.

- *Land Development*: The conversion of rural land to urban land use; it should not be construed to mean conversion to agricultural land uses.

- *Sensitive Wetland*: Wetland types that have a very low tolerance for indirect impacts of stormwater. Examples include bogs, fens, and vernal pools.

- *Stressor*: The negative result of land alteration in a wetland's CDA. Examples include increased water level fluctuations and sediment deposition. Cumulatively, these stressors result in impacts that can change the type of wetland or eventual loss of the wetland.

- *Wetland functions*: Ecological or hydrologic benefits provided by wetlands. Examples include flood attenuation, water quality improvement, wildlife habitat, and groundwater recharge.

- *Wetland services*: Free benefits that result from a wetland's function. This may include benefits such as natural flood control and water treatment, recreation and education, or reduction in damages from storm surges and hurricanes.

Why are Wetlands Important to Watersheds?

Wetlands provide many functions and services within local watersheds, yet the economic benefits of wetlands are seldom appreciated by growing communities. The precise functions and services provided by wetlands depend on their size, type, and location within an urban watershed. Brinson (1993a) developed the HGM (hydrogeomorphic) approach of classifying wetlands based on their hydrologic regimes and landscape position. The HGM classification of wetlands also determines the types of functions provided by the wetland (Table 1). Urban wetlands can improve water quality by removing pollutants, minimizing flood damage by slowing and storing floodwaters, and protecting shorelines from erosion by absorbing storm surges. Wetlands also supply habitat for birds and wildlife and create recreational opportunities. On a global basis, the aggregate value of the ecological services generated by wetlands has been estimated to be $4.9 trillion/year (Costanza *et al.* 1997).

Table 1. Watershed Functions Provided by HGM Wetland Types		
HGM Wetland Type	**Description**	**Common Functions and Values**
Depressional	Topographic depression with closed contours that may have inlets or outlets, or lack them	• Flood storage • Habitat • Pollution treatment • Erosion control
Slope	Surface discharge of groundwater on sloping land that does not accumulate	• Habitat • Pollution prevention • Erosion control
Flat	Low topographic gradients, such as old glacial lake beds, with moderate to abundant rainfall	• Habitat • Pollution prevention • Flood storage • Limited recreational
Riverine	Occur in the floodplain and riparian corridor of larger streams and rivers (e.g., 2^{nd} order and higher)	• Flood conveyance and storage • Shoreline protection and erosion control • Pollution treatment • Fish and waterfowl habitat • Recreation
Fringe	Adjacent to lakes or estuaries	• Habitat • Pollution treatment • Water supply protection (lake fringe only) • Shoreline protection and erosion control • Recreation
Adapted from Kusler (2003), Brinson (1993a), Brinson (1993b), Gwin et al. (1999), and Spivey and Ainslie (no date)		

These "free" services are often taken for granted, but they can easily be lost as wetlands are altered or degraded in a watershed. Richardson (1995) states that when less than 10% of a watershed is wetlands, higher peak stormwater flows will occur. Similarly, Mitsch and Gosselink (2000) estimate that watersheds should retain 3-7% of its area in wetland coverage, at a minimum, to retain adequate flood control and water quality services (Table 2), while phosphorus removal requires as much as 15% wetland coverage per watershed.[1] Preventing the loss of wetland services can be challenging, particularly when financial gains for individual parcel development seemingly outweigh non-market wetland values reaped by the community at large.

However, replacing the lost ecological services of wetlands can be expensive, assuming they can be replaced at all. For example, a community that loses wetland services may need to invest in more costly drinking water treatment, stormwater management, and flood control infrastructure. Similarly, residents may also face higher flood insurance premiums, lower property values, and reduced recreational amenities when wetland services are diminished (Box 2). Communities need to manage wetlands on a watershed basis rather than an individual basis to maximize the watershed value of wetland services.

[1] The percent wetland cover needed in a watershed to maintain these services can vary significantly and depends on factors such as wetland type, watershed size, topography, and area of the country.

Table 2. Estimated Wetland Cover Required in Watersheds for Specific Functions
(from Mitsch and Gosselink, 2000)

Function (*location)*	Watershed area (mi^2)	% Wetland Cover	Reference
Water quality improvement *(IL)*	145	1-5 %	Hay *et al.* (1994)
Phosphorus retention *(Great Lakes Basin, MI)*	80	15%	Wang and Mitsch (1998)
Nitrogen removal *(Sweden)*	341	5%	Arheimer and Wittgren (1994)
Flood control *(Upper Mississippi Basin)*	733,594	7%	Hey and Philippi (1995)
Nitrogen retention *(Mississippi River Basin)*	1,158,306	3.4% - 8.8%	Mitsch *et al.* (1999)

Box 2. Services Lost as Wetland Quality and Function Decrease

As wetlands are degraded or lost within the watershed, communities can expect to pay the price in:

- Decline in water quality as pollutant removal capacity is reduced, triggering additional federal requirements (e.g., TMDLs) for waters not meeting designated use standards
- Increased flood frequency and peak discharges as wetland storage capacity is diminished, resulting in increased property damage, higher flood insurance premiums, and increased public safety concerns
- Increased local costs for infrastructure related to drinking water filtration, stormwater facility maintenance, and flood prevention
- Loss of biodiversity and habitat for aquatic and terrestrial species
- Reduced recreational, educational, and aesthetic open space affecting the quality of life for watershed residents

While a full discussion of the economics of wetlands is beyond the scope of this article, extensive research has been published on the monetary value of wetlands. For a general review of basic wetland valuation techniques, the reader may consult Boyer and Polaski (2004) or Faber and Costanza (1987). The economic value of wetland services can be estimated based on several factors such as:

- The cost to replace wetland services
- What people are willing to pay to enjoy recreational or aesthetic benefits (e.g., travel costs, fees)
- The cost of avoided damages (e.g., flood insurance claims)
- The value of market goods produced (e.g., tons of fish caught)
- The discount or premium in land prices adjacent to wetlands (e.g., home prices adjacent to wetlands)

Estimates of the per acre value of wetland services run as high as $370,000/acre in 1992 dollars (Heimlich *et al.* 1998). The exact value can be attributed to the type and location of the wetland, the services it provides, and the economic methods and assumptions used. More information on the value of various wetland services can be found in Heimlich *et al.* (1998), which summarizes more than 35 wetland valuation studies. Some case studies of the watershed value of wetland services are also presented in Box 3 at the end of this section.

Table 3 summarizes wetland functions and describes the various infrastructure needed to replace those services, most of which are extremely expensive. The next section describes in detail how the following common wetland functions translate into wetland services:

- Pollutant removal
- Flood attenuation
- Groundwater recharge
- Shoreline protection
- Wildlife habitat
- Other services

Table 3. Wetland Functions, Services, and Replacement Options

Function	Services	Alternatives
Pollutant removal	Maintain drinking water quality; process sewage; cycle nutrients; retain sediment; filter runoff; transport organic matter	Water filtration plants; Wastewater treatment plants/ package plants; stormwater facilities with water quality criteria (WQv); Inter-watershed transfer; animal waste storage
Flood attenuation	Storage capacity to reduce downstream flood volume; slow flow to reduce peak discharges and encourage particulates to settle out; protect downstream property; public safety	Stormwater treatment practices (storage); dikes and levees; advanced floodplain construction design
Groundwater recharge and discharge	Maintain baseflow conditions in streams; minimize salt water intrusion	Deeper wells; alternative water source; injection wells
Shoreline protection	Fringe wetlands provide vegetative bank protection; absorb storm surges	Revetments; stream bank stabilization and repair practices; Stormwater treatment practices for channel protection
Wildlife habitat	Habitat for aquatic, terrestrial, and avian species; protective spawning and nursery areas; support biodiversity; biomass production; connective wildlife corridors; habitat for RTE species; foraging grounds for migrating birds	Wetland restoration; species stocking
Other	Recreation, education, and aesthetics (*e.g.*, duck hunting, angling, bird watching, hiking, canoeing, science curriculum, research opportunities, open space, quality of life); commercial products (peat, timber, fish and shellfish, cranberries, and rice)	Wetland restoration

Pollutant Removal

One of the primary watershed functions wetlands provide is maintenance of water quality in lakes, rivers, streams, and groundwater through pollutant removal. Wetlands are natural filters that can remove, retain, or transform a variety of pollutants. Through biological and chemical processes, wetlands intercept surface runoff and remove or assimilate sediment, nutrients, pesticides, metals and other pollutants, and reduce suspended sediment transport (Mitsch and Gosselink, 1993). The actual capability of an individual wetland to remove pollutants is complex and variable. Removal rates vary from wetland to wetland and season to season, and are related to wetland size and type, landscape position, soil properties, groundwater connection, and vegetation among other factors (Mitsch and Gosselink, 1993). In fact, some wetlands can act as sinks for certain pollutants (sediment and nutrients) and sources of others (bacteria). Gabor *et al.* (2004) summarizes key factors that shape nutrient and sediment removal from natural and constructed wetlands.

A few studies have documented the cumulative value of wetlands in reducing watershed sediment loads. Watersheds with more wetland coverage tend to have lower concentrations of suspended solids in receiving waters than watersheds with fewer wetlands (Carter, 1997). In a comparative study in Wisconsin, Novitzki (1979) reported that sediment loads were 90% lower in a watershed with 40% wetland/open water coverage compared to a watershed containing no wetlands.

Nutrient removal by wetlands is important to reduce downstream eutrophication whose symptoms include algal blooms, decreased water clarity, anoxia and fish kills. In freshwater systems, phosphorus frequently causes eutrophication, while nitrogen is more often the culprit in coastal and estuarine watersheds. Wetlands remove nitrogen through settling, denitrification, microbial assimilation and plant uptake. A review of prairie pothole wetlands by Crumpton and Goldsbourough (1998) found that these wetlands were exceptional nitrogen sinks, where denitrification helped reduce nitrogen loads by 80%.

Phosphorus removal is accomplished in several ways in wetlands – settling, adsorption onto organic substrates, precipitation, and biological uptake. Floodplain wetlands have been shown to be particularly good at retaining phosphorus (Craft and Casey, 2000). Mitsch *et al.* (1999) estimated that if the measured phosphorus removal of an Ohio wetland were extrapolated to all of the existing wetlands surrounding Lake Erie, as much as 75-100 tons of phosphorus could be removed annually. Further, Mitsch estimated that if 25% of the original wetland area draining to Lake Erie were to be restored, an additional 24-33% reduction in phosphorus was possible.

Several studies have documented the ability of wetlands to improve the quality of groundwater. A natural marsh wetland was shown to effectively assimilate landfill leachate near Pembroke, Ontario (Fernandes *et al.*, 1996). Richard and Connell (2001) reported reductions in dissolved chlorinated compounds in groundwater for a wetland adjacent to a Minnesota manufacturing site. Wetlands with highly organic substrates and high densities of submerged aquatic plants appear to be able to remove pesticides (Brock *et al.*, 1992). The removal rate for pathogens in natural wetlands has not been widely studied, but research on constructed stormwater and wastewater treatment wetlands indicates that they can be extremely effective. Constructed wetlands designed with long retention times, high light penetration, and emergent vegetation achieved higher pathogen removal rates (Schueler, 1999).

The capacity of wetlands to remove pollutants can be overwhelmed when they receive significant stormwater flows and pollutants from upland development. These stressors alter the chemical and biological processes needed to assimilate nutrients and retain organic matter and sediment. As such, impervious cover in the contributing drainage area can be a strong indicator of the declining performance of wetlands in removing pollutants – more on this topic is provided in Section 3.3. Table 4 summarizes some of the recently published studies on pollutant removal rates for both natural and restored wetlands.

Table 4. Percent Reduction in Common Pollutants in Natural and Restored Wetlands
(Taken from Gabor *et al*., 2004)

Type	Location	Pollutant Type*						
		TN	NO_3	NH_4	TP	SRP	TSS	Pathogens
Natural Wetlands	South Carolina; riparian wetland receiving golf course runoff	n/a	80	n/a	n/a	74	n/a	n/a
	North Carolina; natural wetland receiving stormwater runoff from agricultural land	>80	n/a	n/a	59	y	91	n/a
	China; natural multi-depression wetland system receiving continuous surface runoff	n/a	n/a	n/a	93.9	90.0	94.9	n/a
Restored Wetlands	Spain; restored wetland receiving rice field runoff	50 to 98	n/a	n/a	n/a	<50	n/a	n/a
	Maryland; restored wetland in agricultural watershed (two year average)	n/a	35	25	n/a	n/a	0	n/a
	Georgia; restored riparian wetland adjacent to manure application area	n/a	78	52	66	66	n/a	n/a
	Alberta; restored marsh receiving wastewater; summer	n/a	87	76	64	n/a	n/a	n/a
	Alberta; restored marsh receiving wastewater; winter	n/a	-26	46	n/a	26	n/a	n/a
	Maryland; restored wetland receiving agricultural runoff	n/a	68	n/a	43	n/a	n/a	n/a
	Illinois; restored wetland receiving agricultural runoff	n/a	36 to 45	n/a	20	n/a	n/a	n/a

* TN = Total Nitrogen; NO_3 = Nitrate; NH_4 = Ammonia; TP = Total Phosphorus; SRP = Soluble Reactive Phosphorus; TSS = Total Suspended Solids

Flood Attenuation

Wetlands have the potential to collect, store, and slowly release runoff and floodwaters gradually over time. The degree of flood control depends on the size and shape of the wetland, its landscape position, the depth to the water table, soil permeability and slope. The storage function helps to minimize flooding of downstream properties, slow erosive flows in stream channels, and delay the arrival of peak discharges. For example, one study indicates that wetlands can store almost all of the snowmelt runoff generated in their watersheds, which can be very important in regions of the country where snowmelt flooding is a concern (Hayashi *et al.*, 2003). The value of wetland flood storage is often greatest in urban watersheds where past development has sharply increased peak discharges during flood events.

Filling depressional wetlands and encroaching onto floodplain wetlands can reduce their capacity to attenuate flooding in a watershed. For example, Gosselink *et al.* (1981) reported that the loss of floodplain forested wetlands and confinement by levees has reduced the floodwater storage capacity of the Mississippi River by 80%. When the wetland flood control function is diminished, most communities resort to a patchwork of expensive engineering practices to fix recurring flooding problems such as floodways, channelization, stormwater detention ponds, and levees. These engineering "fixes" are not always effective, particularly during extreme storm events such as the Mississippi floods in 1997.

The ability of wetlands to attenuate floods is evident when a hydrograph from a watershed with extensive wetlands is compared to a watershed with fewer remaining wetlands (Figure 1). The dramatic flood attenuation depicted for the Charles River (MA) was used to justify the purchase of 8,500 acres of headwater wetlands in the watershed as a cost-effective and natural flood control strategy. In this case, the 8 million dollar cost to preserve the wetlands was far less than $30 million price tag to construct engineered flood control structure to prevent flood damages (Thibodeau and Ostro, 1981).

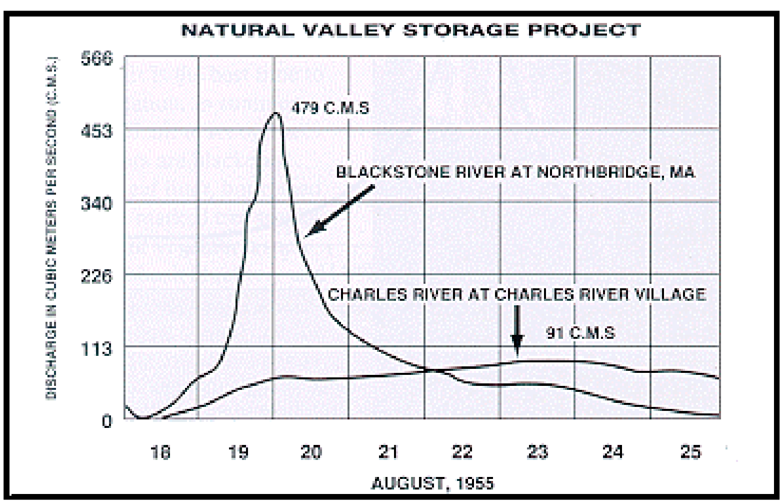

Figure 1. Hydrograph showing peak flows for the Charles River Watershed as compared to the Blackstone, which is a similar watershed with fewer remaining wetlands. Presented by Welsch et al. (1995) as derived from Thibodeau and Ostro (1981)

Groundwater Recharge and Discharge

Some wetlands help maintain water table levels by recharging groundwater, while other wetlands serve as discharge points for groundwater and thus provide a water source for adjacent wetlands and/or maintain baseflow discharges to streams (O'Brien, 1988 and Winter, 1988). These functions may be of particular importance when residents rely on groundwater as the source of their drinking water. In 1995, the U.S. Geological Survey (USGS) reported that groundwater supplied drinking water for 46% of the nation's overall population and 99% of the population in rural areas (in USEPA, 2002c).

The interaction between an individual wetland and local groundwater is complex and depends to a great extent on its landscape position. In addition, the magnitude of groundwater recharge rates is related to underlying soil permeability, vegetation density, and water table gradient (Carter and Novitzki, 1988; Weller, 1981). Peatlands, for example, are often separated from underlying groundwater by an impermeable layer of rock or soil. In these situations, groundwater recharge typically occurs through soils found around the perimeter of the wetland (Verry and Timmons, 1982).

Olewiler (2004) reports that forested wetlands overlying permeable soils can release up to 100,000 gallons per acre each day into groundwater. Groundwater levels can be greatly affected if wetlands are drained or developed. For example, a study by Ewel (1990) on a cypress swamp in FL suggests that if 80% of the wetland were drained, associated groundwater would be reduced by 45%.

Shoreline Protection

Because of their position in the landscape, fringe wetlands have the capacity to protect streams and shorelines from erosive winds, waves, and currents. These fringe wetlands contain vegetation and roots that consolidate soils, absorb wave energy, and help dissipate surface flows. The capacity of a wetland to dissipate erosive forces depends upon vegetative density and root structure, soil type, and the frequency and intensity that waves meet the shore or runoff cuts the bank. Wetland vegetation also increases local sedimentation, which can help build up streambanks and shorelines over time. When wetland vegetation is removed, increased shoreline and bank erosion can occur. This can result in loss of property, threatened infrastructure, or public safety concerns. States such as Florida and Louisiana are restoring coastal wetlands to serve as a buffer between development and hurricane storm surges.

When the shoreline protection services of wetlands are lost, the costs to stabilize eroding stream banks and shorelines rise sharply. The precise cost depends on the length stabilized, the types of engineering techniques employed (hard or soft), and site access. Large-scale projects such as beach renourishment, marsh restoration, and construction of revetment or bulkheads can be extremely expensive, require ongoing maintenance, and involve numerous state and federal permits. Even soft shoreline restoration projects are not inexpensive. Figure 2 depicts a fringe wetland restoration project along 175 feet of private shoreline. By adopting and enforcing shoreline criteria to fringe wetlands and their buffers, many of these costs can be minimized or avoided (see Article 3).

Figure 2. Before (top) and after (bottom) shots of a small shoreline restoration project. Here wetland plantings and rock sills were combined to prevent shoreline erosion. (Photos courtesy of South River Federation)

Wildlife Habitat

Comer *et al.* (2005) reported that wetlands provide habitat for more aquatic, terrestrial, and avian species on an area basis than any other habitat type, making them one of the most ecologically and economically important ecosystems on earth. In fact, more than 35,000 rare plants and animals are found in wetlands in the U.S. alone (Comer et al., 2005). Almost half of all federally threatened and endangered species rely on wetlands directly or indirectly during some stage of their life cycle, and all federally listed amphibian, fish, clam, and crustacean species depend on wetlands (NatureServe, 2003).

Wetlands produce considerable biomass and contain a mosaic of upland and wetland habitat features that helps support their high biodiversity. Each wetland type offers a unique mix of habitat elements such as cover, food, water, nesting and other life sustaining features. Many bird species rely on a variety of wetland types for foraging, breeding, and nesting habitat, particularly migrating waterfowl. For example, black ducks that winter in the coastal wetlands of the Chesapeake Bay rely on the prairie potholes in the upper mid-west for nesting. Coastal wetlands also serve as important nursery grounds for many recreationally and commercially important fish and shellfish, including shrimp. Some riparian and riverine wetlands in the Pacific Northwest are critical in the lifecycle of listed salmon runs.

Wetlands often form the remaining structure of wildlife corridors and open space in urban watersheds. The corridor function can be degraded, as shown by direct and indirect wetland impacts that commonly occur in urban watersheds. As will be documented in Section 2, changes in wetland hydrology from upland development can have significant consequences on downstream aquatic species (Carter, 1997; Owen, 1999 and Kercher *et al.*, 2004).

Even degraded urban wetlands can still provide some degree of habitat function. For example, the Hackensack Meadowlands at one time were dominated by Atlantic white cedar swamps, salt marshes, and other wetland habitats. After centuries of development, these urban wetlands are now fragmented by dikes, fill and landfills and are extensively crossed by highways, railroads, and pipelines. Despite the cumulative loss of wetlands and the degradation of remaining wetlands, the Meadowlands still attract migrant and breeding waterfowl, are home to a moderate diversity of fish and other animals, and are host to a few rare plants (Kiviat, 2004).

Restoring degraded wetland habitat functions can be difficult and costly and can take many years. Some wetland types are more easily restored than others (fringe and open water wetlands versus bogs and fens). Wetland restoration may also never successfully replicate the original wetland community, particularly if hydrologic and water quality stressors in the contributing drainage area are not effectively managed.

Other Wetland Services

Wetlands provide many other services that generate a tangible economic benefit to communities including recreation, natural resources and education. For example, more than half of all adults hunt, fish, bird watch, boat, or photograph wildlife annually; much of this activity is centered around wetlands. In fact, more than 82 million Americans spent more than $108 billion on these activities in 2001 (USFWS, 2002).

Hunting and fishing are popular activities enjoyed by millions every year. An estimated $600 million is spent annually by waterfowl hunters on wetland bird species (USEPA, 1995). In 2001, nearly 10 million hunters spent more than $2.2 billion in pursuit of migratory birds and small mammals including muskrats and beaver that are often found in wetlands (USEPA, 2006). In 2004, harvested muskrat pelts were estimated to be worth about $124 million (USEPA, 2006).

The value of crab, shrimp and salmon in the U.S. was estimated at $1.167 trillion in 2004, all of which depend on wetlands for part of their life cycle. In fact, as much as 75% of commercially-harvested fish and shellfish and up to 90% of recreational fish in the U.S. rely on wetlands for some or all of their life cycles. (USEPA, 2006). A recent survey found that anglers spent an estimated $14.7 billion for fishing trips, $17 billion for equipment, and $4 billion in miscellaneous costs (e.g., licenses, stamps, land, magazines, etc.) in 2001 (USFWS, 2002).

As noted previously, the cost to replace lost wetland services is typically much higher than the cost of wetland protection. Box 3 reviews selected case studies that estimate the value of wetland services and/or estimate the cost to replace them with engineered practices.

The many functions and services provided by wetlands can be lost by both direct and indirect impacts to wetlands. The remainder of this article defines and describes direct impacts to wetlands and introduces the regulations that govern a portion of these impacts. The indirect impacts resulting from land development in the contributing drainage area are discussed. Studies that document the effects of these changes on wetlands are discussed, and the ultimate effects on wetland quality throughout a watershed are identified.

Box 3. Case Studies on the Value of Wetland Services and Costs to Replace Them

Pollutant Removal

- Replacing natural water filtration services such as wetlands is costly. More than $200 million is required to construct filtration plants and even more to operate (Barclay *et al.,* 2004). New York spent $1.4 billion on watershed protection for Catskills drinking water supply to avoid estimated filtration plant construction costs of $4-6 million dollars with annual operating costs of $300,000. To avoid building and operating a $200 million artificial water filtration plant, Portland, Oregon, spends $920,000 annually to protect the watershed (Krieger, 2001).
- Annual nitrogen and phosphorus waste treatment benefits received from existing 100,000 acres of wetlands in Lower Fraser Valley, British Columbia were estimated at $18 to $50 million dollars per year (Olewiler, 2004)
- Breaux *et al.* (1995) estimated annual wastewater treatment costs savings of $6,000-$10,000 per wetland acre at 15 Louisiana seafood processing plants (1992 dollars).

Flood Control

- The current flood protection benefits of wetlands in two Washington cities ranged between $36,000-$51,000 per acre (Leschine et al. 1997). The value of wetland flood control services increases as wetlands become more fragmented in urban watersheds.
- The cost to replace the flood control function of 5,000 acres of drained wetlands in Minnesota was found to be $1.5 million annually or about $300 to replace each acre/foot of flood water storage (Sipple, 2002).

Groundwater Recharge

- The value of a 550,000 acre swamp in Florida for aquifer recharge and flood storage was estimated to be $25 million/year (DU, no date)
- Acharya (2000) estimated the value of groundwater recharge for dry season agricultural irrigation and domestic use of the Hadejia-Nguru wetlands in northern Nigeria to be 6% of the yearly income per farmer.

Habitat

- More than $100 million in state and federal funds has been allocated for salmon recovery and habitat protection in the Pacific Northwest between 2000-2003 (Barclay *et al*., 2004).
- Residents of Tillamook, Oregon, value additional salmon habitat at $5,000/acre (Gregory and Wellman, 2001).
- Imus (2003) presents estimated project costs for wetland habitat restoration ranging from $50,000 to $300,000/per acre for projects under 50 acres.

Direct Impacts to Wetlands

Direct impacts occur when a wetland is dredged, filled, drained or otherwise altered by activities occurring inside the wetland boundary. Most direct wetland impacts are regulated to some degree under federal, state, and local wetland permit programs. Examples of direct impacts include draining wetlands for agricultural use by constructing drainage ditches or installing underground drainage tiles (Figure 3) and filling wetlands to provide useable land on which to build (Figure 4). Direct impacts usually result in wetland loss.

Figure 3. Drainage tiles are installed to convert wetlands to useable land for agriculture, a major cause of wetland loss (Source: Michigan State University Department of Geography)

Figure 4. A portion of this wooded swamp (left) was filled to build a new subdivision in the Buzzard's Bay watershed, Massachusetts (Source: Buzzard's Bay National Estuary Program)

National Estimates of Wetland Loss from Direct Impacts

Dahl (1990) estimated that 221 million acres of wetlands existed in the lower 48 states in 1780. In the two centuries since then, more than 53% of wetland cover has been lost due to draining, dredging, filling, and flooding, which equates to a loss rate of 60 acres of wetlands per hour. Wetland loss has slowed considerably in the last two decades due to federal and state wetland permitting and increased wetland restoration (Table 5).

Table 5. Summary of Wetland Loss from the 1950s through 2004

Time Period	Net Loss of Wetlands	Net Annual Rate of Loss	Types Lost	Major Causes of Loss	Source
1950s to 1970s	9.1 million acres	458,000 acres per year	Majority of losses were freshwater wetlands	Agriculture (87%) Urban Development (8%) Other (5%)	Frayer *et al.* (1983)
Mid-1970s to mid-1980s	2.6 million acres	290,000 acres per year	98% of losses were freshwater wetlands	Agriculture (54%) Other* (41%) Urban Development (5%)	Dahl and Johnson (1991)
1986 to 1997	644,000 acres	58,500 acres per year	98% of losses were freshwater wetlands	Urban and Rural Development (51%) Agriculture (26%) Silviculture (23%)	Dahl (2000)
1998 to 2004	Net gain of 191,000 acres	Net gain of 32,000 acres	Net gain due to creation of 695,400 acres of ponds	Urban and Rural Development (61%) Agriculture (17% increase) Silviculture (8%)	Dahl (2006)
*Wetlands that have been cleared and drained but not yet put to a definable use					

The U.S. Fish and Wildlife Service tracks the status and trends in wetland loss in the conterminous United States at regular intervals. Wetlands trend data are derived from remote sensing data and field surveys of randomly selected sample plots (Dahl, 2000 and Dahl, 2006). It is important to keep in mind that national wetland tracking efforts are subject to some limitations that may cause wetland loss to be underestimated. For example, the National Wetland Inventory (NWI) methods exclude very small or ephemeral wetlands and do not easily distinguish losses by wetland type, quality, or function (see Box 4).

Dahl (2000) estimates that 105.5 million acres of wetlands were present in 1997, which comprises about 5.5% of the land surface of the lower 48 states. Historically, most wetland loss has occurred in freshwater wetlands. Loss of estuarine wetlands has been smaller, but also reflects the fact that estuarine wetlands comprise less than 5% of the national total (Figure 5).

Box 4. Concerns About National Estimates of Wetland Loss

- Data collection methods used by USFWS have improved in accuracy over the years, allowing the NWI to capture smaller wetlands or wetland types that would not have been included in previous inventories. This may result in an overestimate of wetland gains.
- USFWS methods do not include ephemeral wetlands (wetlands that are dry for some portion of the year), wetlands smaller than one to three acres, Pacific coast estuarine wetlands, or wetlands that were previously converted for agricultural use (Dahl, 2000). The exclusion of ephemeral wetlands is particularly troublesome because they are often the most vulnerable to wetland loss since they are easily converted for development or agriculture. The NWI also does not effectively capture narrow, riverine wetlands and forested wetlands. (http://www.ag.iastate.edu/centers/iawetlands/NWIhome.html)
- USFWS reports do not consider the *quality* of the wetlands in question. For example, open water ponds were found to have the largest gain in area of all freshwater wetland types, probably due to creation of retention ponds to treat runoff from new developments (Dahl, 2006). Although the acreage of these ponds has increased, they arguably do not provide the same functions as other wetland types and therefore cannot be used to replace natural wetlands lost to direct impacts. Similarly, wetland creation, mitigation and restoration efforts that counted as gains may not replicate the quality or functions of the original wetlands lost.
- The watershed is generally a more accurate scale to track losses and gains in wetland acreage and functions than national estimates.

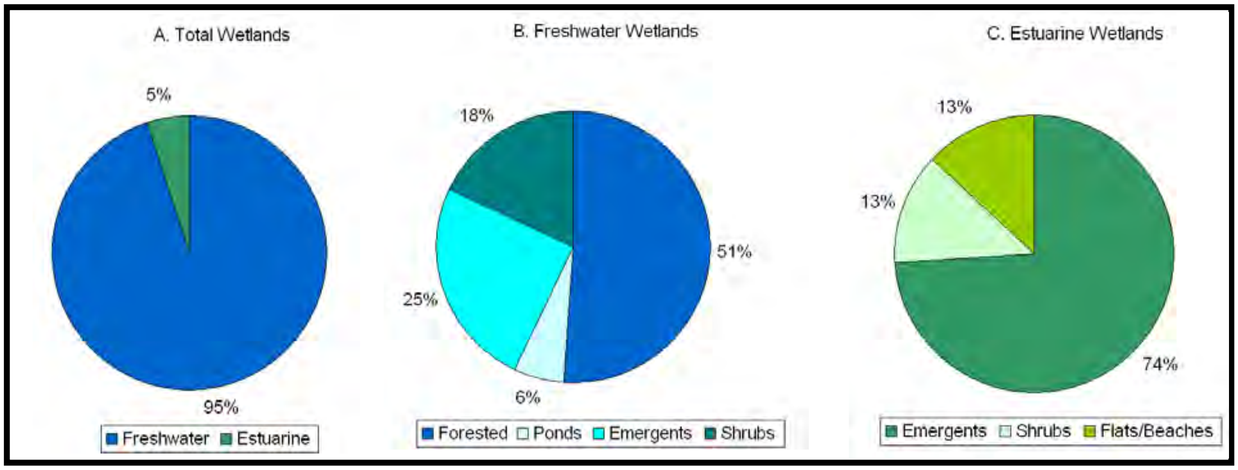

Figure 5. Distribution of Wetland Types in the U.S. (Source: Dahl, 2000)

As was shown in Table 5, the annual rate of wetland loss has decreased sharply, and a slight gain in wetland cover was actually reported in the most recent survey period (1998 to 2004). This slowdown in wetland loss may reflect more active enforcement of federal, state, and local wetland permits, and changing perceptions about wetland values and benefits. In addition, wetland area losses have been partially offset by gains in the creation of new wetlands in agricultural areas. Farm Bill programs such as the Wetlands Reserve, Conservation Reserve, Farmed Wetland Option and Conservation Reserve Enhancement have resulted in the creation of extensive amounts of freshwater wetlands. Dahl (2006) estimates that 564,300 acres of wetlands on agricultural lands were created from 1998 to 2004. This was offset by 488,200 acres of loss due to agricultural land use[2] during the same period, resulting in a net gain of 76,100 acres.

Two other key findings emerged from the most recent tracking survey. First, urban and rural development now account for more than 60% of national freshwater wetland loss with the balance lost to silviculture and conversion to deepwater habitat. The share of wetland loss due to development has steadily risen in each reporting period. Most of the losses due to land development occur in rather small increments- usually less than five acres at a time. The finding that development is now the leading cause of wetland loss argues for a stronger role for local governments to protect wetlands from the direct and indirect impacts of land development.

The second key finding is that nearly 700,000 acres of ponds were created from 1998 to 2004 period that are now classified as wetlands. Without these gains due to ponds, national net loss of wetlands would have continued in the last reporting period. The surge in pond creation was attributed to construction of ponds on agricultural as well as non-agricultural lands and may also reflect the construction of stormwater ponds in many parts of the country to comply with new state and federal stormwater rules. Most wetland scientists, however, do not consider ponds to have the same functional quality as a natural wetland. Indeed, research profiled in Section 3 argues that most stormwater ponds and wetlands are particularly degraded systems. This finding

[2] Agricultural land uses included horticultural crops, row and close grown crops, hayland, pastureland, native pastures and range land and farm infrastructures.

underscores the need for communities to go beyond the goal of no net loss in wetland acreage and seek to maintain overall wetland quality and function. Watersheds are often the most accurate and appropriate geographic unit to track losses and gains in wetland acreage and functions at the local scale.

Federal Programs to Reduce Direct Impacts to Urban Wetlands

Direct impacts to wetlands have historically been regulated by the federal government, although states, tribes, and local governments also have been involved. Tiner (1984) estimates that only about 0.5% of privately-owned wetlands in the lower 48 states are under some form of conservation easement or protection. The remaining 99.5% are potentially subject to direct impacts.

The primary federal authority to protect wetlands from the direct impact of land development is Section 404 of the Clean Water Act (CWA). Section 404 regulates the discharge of dredged or fill material into waters of the U.S., including wetlands. A Section 404 permit must be secured from the U.S. Army Corps of Engineers (USACE) or a delegated state agency before any of these activities can be undertaken. The permit applicant must demonstrate that they have taken steps to avoid impacts to a wetland, minimized any potential impacts, and performed mitigation to compensate for any unavoidable wetland impacts, to the extent practicable. Compensation can be provided by restoring a former wetland, enhancing a degraded wetland, creating a new wetland, or more rarely, preserving an existing wetland. The USACE administers the Section 404 wetland permit program with support from the U.S. Environmental Protection Agency (USEPA). USFWS and the National Marine Fisheries Service serve in an advisory capacity.

An *individual* permit is required for potentially significant wetland impacts, which are reviewed individually by the USACE. However, a *general* permit can be issued for discharges to wetlands that are deemed to only have a minimal adverse impact. General permits are issued on a nationwide, regional, or state basis for particular categories of activities, such as minor road repairs, utility line backfill, and surveying within wetlands. Nearly 85% of all wetland permit applications fall under *general* rather than *individual* permits (Davis, 1997).

Activities covered by general permits do not require public notice, and the general public and state and federal agencies do not have the opportunity to comment on individual projects. In some instances, the general permit process may eliminate individual review and allow certain activities to proceed with little or no delay, provided that applicants meet specific conditions outlined in the general permit. Under some nationwide general permits, permittees can proceed with activities without notifying the USACE (e.g., projects that disturb less than a specific acreage threshold) and are not required to perform mitigation (NRC, 2001). General permits cover projects thought to have minimal adverse environmental impacts, such as minor road crossings. As can be seen in Figure 6, the term “minimal” is subjective and is not defined in the CWA, so it is clearly subject to broad interpretation by the USACE and other review authorities.

The Section 404 wetland permit program does not regulate all activities in all wetlands. Section 404 exempts certain activities that may result in direct impacts to wetlands, such as farming, silviculture, and ranching activities that are part of an established and ongoing operation, as well as certain maintenance and construction activities. The Section 404 program only regulates disposal of dredge or fill material into wetlands, but does not always regulate other activities, such as removal of material, drainage, or removal of vegetation. Box 5 outlines the limitations of the Section 404 program in protecting wetlands from direct and indirect impacts.

The scope of the 404 permit program has been profoundly influenced by several recent Supreme Court decisions. For example, in a 2001 case, the so-called SWANCC ruling (Solid Waste Agency of Northern Cook County) potentially reduced the acreage of wetlands subject to Section 404 permits. SWANCC appealed the denial of a Section 404 permit to fill an abandoned sand and gravel pit that had turned into a wetland and was being used by migratory birds. The Supreme Court ruled that the Corps of Engineers could not deny a Section 404 permit to alter isolated wetlands and other waters based on use by migratory waterfowl alone. Because the Supreme Court did not clearly define what was meant by "isolated" waters, it has been left to individual lower courts to decide which wetlands are still subject to the 404 permit program. According to some estimates, as many as 20 million acres of wetlands are at risk due to the SWANCC ruling – approximately 20% of the nation's wetland inventory (NRDC, 2003; Figure 7).

Figure 6. Wetland fills for road crossings such as this one in Minnesota are often covered under general permits
(Source: Minnesotans for Responsible Recreation)

Box 5. Limitations of the Section 404 Program

- Does not protect wetlands from indirect impacts that occur within wetland contributing drainage areas (e.g., increased stormwater runoff or pollutant loads).
- Some isolated wetlands may be outside the geographic jurisdiction of the program.
- Some activities are not subject to regulation (e.g., drainage; removal of vegetation; activities that result in only incidental fallback; normal agricultural, silvicultural or ranching activities that are part of an established and ongoing operation).
- Most activities that are subject to regulations are authorized by general permits, which do not have as extensive a review process and may not require any mitigation.
- Does not address cumulative impacts to wetlands due to the permit-by-permit approach as opposed to a watershed approach.
- Does not successfully replace wetland types or functions because mitigation wetlands are often not of the same type as the wetland they are replacing, and insufficient guidance exists on how to mitigate for functions and measure success.
- Does not always replace lost wetland acreage due to high failure rates of mitigation wetlands or lack of implementation and enforcement.

Figure 7. Prairie pothole wetlands such as these in South Dakota are generally considered isolated and may be vulnerable to direct impacts in light of the SWANCC ruling (Source: NRCS photo gallery)

In June 2006, the Supreme Court ruling in *Rapanos v. U.S.* and *Carabell v. U.S. Army Corps of Engineers*, potentially increased the vulnerability of many smaller streams and wetlands. In these cases, the Corps sued two Michigan property owners (Rapanos and Carabell) for filling in some infrequently saturated wetlands. The Court's split decision ultimately sent the cases back to the lower courts to decide, but the controlling opinion was that in order to assert jurisdiction over such waters, the Corps would have to establish a "significant connection" between wetlands and "navigable waters" on a case-by-case basis.

Because the Court did not define what constitutes a "significant connection," this ruling, along with the SWANCC decision, may leave isolated wetlands and ephemeral and intermittent streams vulnerable to loss under the Clean Water Act (e.g., Section 404 permits, National Pollutant Discharge Elimination System, and Section 401 Water Quality Certification). Article 6 provides further discussion of this issue.

Another wetland enforcement issue involves what is known as the "Tulloch Rule," which allowed developers to ditch or drain wetlands, provided the activity did not result in a redeposit of material back into the wetland (referred to as "incidental fallback"). In 2001, USEPA and USACE attempted to reduce wetland losses from this loophole by defining specific activities, such as ditching, draining, in-stream mining, and channelization that cause more than incidental fallback and are therefore subject to 404 regulation, unless project-specific evidence shows otherwise. As can be seen in Figure 8, ditching under the Tulloch Rule still continues in some regions of the country.

Figure 8. This South Carolina wetland was being drained and ditched in July 2006.

Another limitation of the Section 404 permit program occurs when mitigation is used to compensate for unavoidable wetland loss. Two independent reports conducted in 2001 by the National Research Council (NRC) and the U.S. Government Accountability Office (GAO) questioned the effectiveness of mitigation efforts. The GAO report focused on mitigation that occurred under an in-lieu-fee arrangement, where developers pay fees to public or private wetland banks that fund creation or restoration wetlands. The GAO report concluded that "the extent to which the in-lieu-fee option has achieved its purpose of mitigating adverse impacts to wetlands is uncertain." No data was available to determine whether wetland banks were actually mitigating wetland losses, and in some cases, whether required mitigation was ever performed. Some USACE district offices considered mitigation to be successful as soon as the developer wrote a check, even if no mitigation was performed.

NRC (2001) concluded that the goal of no net loss of wetlands was not being met from the standpoint of wetland function. A review of Section 404 wetland permits issued from 1993 to 2000 indicated that about 24,000 acres of wetlands were permitted to be filled each year at the same time that 42,000 wetland acres were created as part of compensatory mitigation, yielding a net gain in wetland acreage on paper. NRC found, however, that data was inadequate to determine if mitigation was actually successful in replacing lost functions because wetland functions lost due to permitted fills were never reported. In other cases, the review found that as many as 34% of required mitigation projects failed to meet permit conditions or were not implemented.

Indirect Impacts to Urban Wetlands

Indirect impacts are caused by increased stormwater and pollutants generated by land development within a wetland's contributing drainage area (CDA) that stress the plant and animal community. Because wetlands are often located at the topographic low point of a watershed, they are often profoundly influenced by activity in upland areas. It is important to note that most federal, state, and local wetland permit programs start and stop at the wetland boundary and do not consider or regulate activities that occur within wetland CDAs.

This section reviews research on the indirect impacts to wetlands in three steps. First, research is profiled that documents changes in hydrology as land development occurs in the CDA. The next part reviews research that shows how these factors alter the hydrologic conditions and water quality within urban wetlands. Lastly, the research on how plants and animals respond to these urban wetland stressors is summarized.

Finally, the section concludes with a review of the research on the cumulative impact of land development on wetland plant, aquatic invertebrate, amphibian, bird, and mammal communities. Based on this review, the section examines whether it is possible to predict biological response based on a CDA metric such as impervious cover. Although impacts are frequently seen at low levels of development, there is not enough comparable data to warrant an Impervious Cover

Model[3] approach at this time. Instead, the concept of sensitive wetlands is introduced as a management alternative, and a preliminary list of sensitive wetland communities is presented.

Hydrologic Changes in the Contributing Drainage Area

Three main processes associated with land development significantly change the hydrology of the CDA. First, native vegetation that once intercepted rainfall is removed and soils are compacted. Second, impervious cover is created when roads, rooftops, and parking lots are constructed, which greatly increases runoff volumes. Lastly, efficient storm drainage systems are installed to quickly convey runoff to downstream waters, including wetlands. As a result of these changes, infiltration and recharge of groundwater is diminished.

The construction of roads across streams and wetlands can also cause hydrologic changes that extend a significant distance upstream and/or downstream. Crossings of an individual wetland can cause direct wetland impacts, which may be regulated under Section 401 or 404 of the CWA. However, wetlands can also be indirectly impacted by roads that cross the wetland, tributaries to the wetland in the CDA, or just downstream of the wetland. The primary indirect impact is flow constriction.

The three changes from land development with the most potential to impact wetlands include:

- Increased stormwater runoff
- Decreased groundwater recharge
- Flow constrictions

Stormwater runoff to wetlands is strongly influenced by impervious cover within the CDA and may increase annual surface runoff to a wetland by one to two orders of magnitude.

This section describes how land development contributes to each of these hydrologic changes. The following section then describes how these changes result in specific hydrologic stressors to wetlands. Kercher and Zedler (2004) summarize a wide range of studies that indicate how land development in the CDA alters the natural hydrologic regime of wetlands.

Increased Stormwater Runoff

Stormwater engineers have shown how increases in impervious cover in a watershed can dramatically increase the rate and volume of stormwater runoff compared to pre-development levels (Schueler, 1987). Stormwater runoff can increase by one to two orders of magnitude, depending on the nature of the predevelopment land cover (Schueler, 1987). The predictable increase in runoff generation as a function of impervious cover is illustrated in Figure 9. Depending on how much of the CDA is covered by impervious surfaces, stormwater runoff to a downgradient wetland can increase dramatically -- particularly if stormwater is directly discharged to a wetland through a ditch, channel or storm drain pipe.

For example, Schueler (2001a) reported that the total runoff volume from a one-acre parking lot is about 16 times greater than that produced by an undeveloped meadow. Cappiella *et al.* (2005)

[3] The Impervious Cover Model (ICM) illustrates the relationship between subwatershed IC and expected stream quality, and defines three broad urban subwatershed categories–impacted streams, non-supporting streams, and urban drainage. For additional information on the ICM, see Schueler (2004).

noted that the same sized parking lot generates 19 times more runoff than an acre of mature forest. Table 6 shows the changes in the runoff coefficient, which expresses the fraction of annual rainfall converted to stormwater runoff, for forest, turf and impervious cover. It is important to note that the impact of increased stormwater runoff largely depends on the wetland's landscape position. For example, fringe wetlands in lacustrine, riverine or estuarine settings tend to be impacted less because they are more affected by water levels in the adjacent lake, estuary or river than by local surface runoff in the watershed.

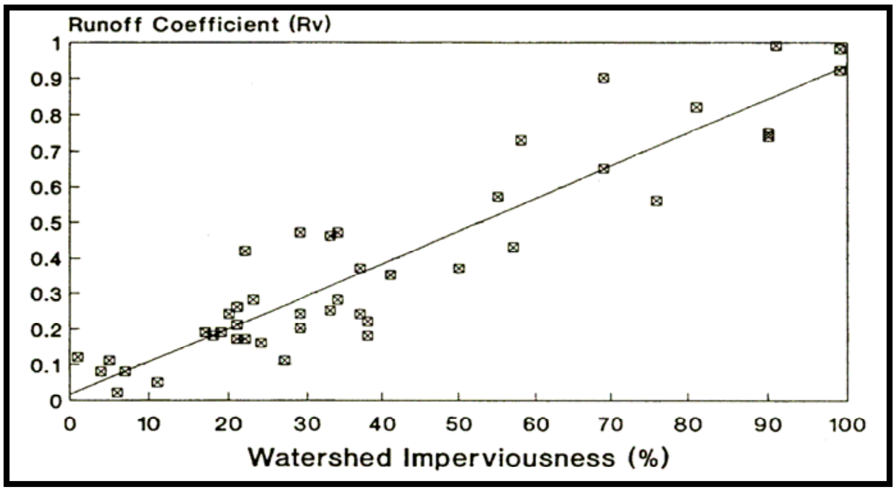

Figure 9. Increased runoff coefficients with changes in watershed imperviousness
Source: Schueler, 2000

Table 6. Runoff Coefficients for Various Land Uses		
Forest Cover [1]	**Turf Cover** [2]	**Impervious Cover**[3]
0.05	0.10	0.95
1: Measured runoff coefficient from Mostaghimi, *et al.* (1994). 2: Average from Legg, *et al*. (1996) and Pitt (1987) for B and C soil types. 3: Regression of 40 sites nationally in Schueler (1987).		

Decreased Groundwater Recharge

As impervious cover increases within the wetland CDA, infiltration of rainfall into the soil is proportionately reduced. Compaction of upland soils during the construction process can also sharply reduce soil infiltration rates (Schueler, 2001). This reduced infiltration translates into reduced groundwater recharge that is needed to sustain many wetlands. Numerous studies have also shown that impervious cover can reduce groundwater recharge in small urban watersheds (e.g., CWP, 2003 and Saravanapavan *et al*., 2004).

Flow Constrictions

Flow constrictions can be caused by the construction of roads, bridges, pipelines or other structures across individual wetlands, or upstream or downstream of them (Figure 10). Perhaps the most common cause of flow constriction is when culverts are installed to provide a conduit to move water underneath a road. Although most culverts are sized to carry flow from 10 to 100-year recurrence design storms, they often lose hydraulic capacity due to sedimentation and increased peak flows from new upstream development. Undersized culverts cannot fully convey the increased flows from the watershed and create a constriction to flow that can impact the hydrology of both upstream and downstream wetlands. This disconnection is often very significant for tidal wetlands as the crossing impedes the natural flux of water from storms and tides.

While much research has focused on the hydrologic impact of larger road and highway crossings (see Richardson *et al.*, 2003; Forman and Deblinger, 2000; Richardson and Nunnery, 1998), smaller, local road crossings are much more common in the urbanized landscape and may exert a stronger hydrologic impact (Gailbrath *et al.* 2005). May *et al*. (2000) observed that road density in urban watersheds is a direct function of impervious cover, and Schueler (2004) reported the density in urban watersheds at two to 10 crossings per stream mile. The high road and stream crossing density observed in urban watersheds suggests that there is a high probability that many urban wetlands will experience flow constriction to some degree.

Researchers have documented the upstream and downstream hydrologic impacts of crossings on wetlands, and the high crossing density expected in most urban watersheds suggests that constrictions may be a common problem for many urban wetlands.

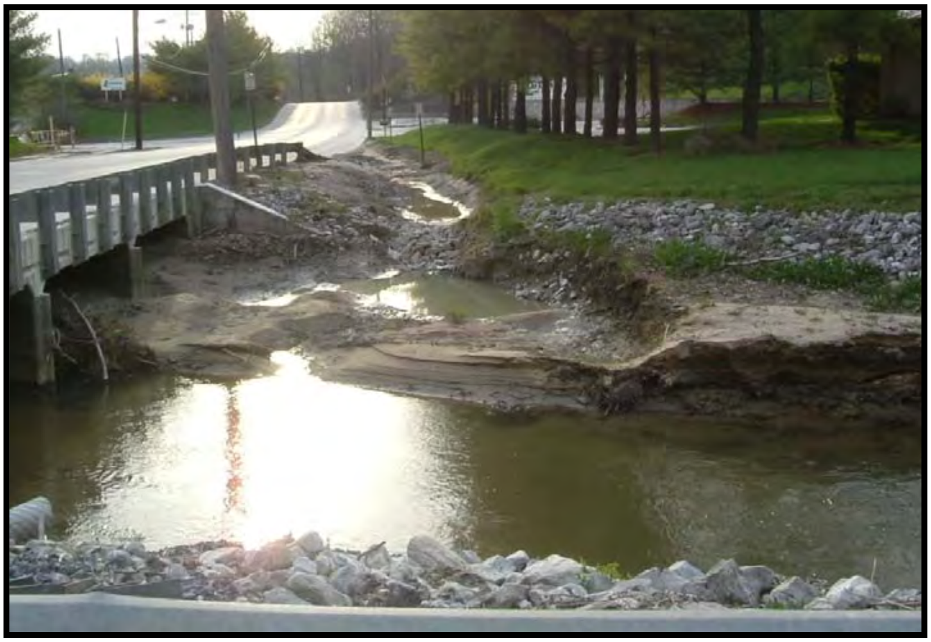

Figure 10. Road crossings interrupt urban streams and can affect hydrology of streams and wetlands.

Hydrologic Stressors to Urban Wetlands

Wetland scientists consistently regard hydrology as a critical factor in determining the type and functions of a wetland and the kinds of plants and animals that will inhabit it (Welsh, *et al.*, 1995; Mitsch and Gosselink, 1993). Hydrology is one of the three defining factors to delineate a wetland and is used in most wetland classification schemes. The natural hydrology of a wetland evolves in response to groundwater contributions, adjacent surface waters, and gains and losses from surface flows, precipitation, and evapotranspiration.

The previous section demonstrated how development in the CDA can alter many of the variables of a wetland water balance. The key question is at what point do these hydrologic changes create significant wetland alterations that stress the plant or animal community? Researchers have reported five possible ways that changes in CDA hydrology can become a stressor within an urban wetland:

1. Increased ponding
2. Increased water level fluctuation
3. Flow constrictions
4. Decreased groundwater discharge
5. Hydrologic drought in riparian wetlands

Each hydrologic stressor and its resulting impacts on the plant and animal community are described below.

Increased Ponding

Increased stormwater runoff can lead to increased ponding depth within a wetland, particularly if the wetland has a water budget that was previously dominated by precipitation, or if it does not have the capability to rapidly discharge excess stormwater runoff via groundwater, surface water discharge, evapotranspiration, or other outlets. When this occurs, there inevitably will be deeper water in the wetland throughout much or all of the year. Increased ponding can also be caused by constrictions below wetland outlets such as undersized culverts.

Table 7 summarizes the rather limited research on the effects of increased ponding on wetlands. An excellent example of the ponding effect is reported by Owen (1999) who analyzed historical changes in water depth in a wetland adjacent to the Yahara River in Wisconsin. By 1990, the wetland's watershed had become 63% impervious, and stormwater runoff was conveyed to the wetland through ditches. Increased stormwater runoff due to land development produced a 20-fold increase in surface depth in the wetland from levels measured in 1850. The increased ponding promoted conditions favoring the spread of invasive wetland plants. Ernst and Brooks (2003) observed that increased ponding in forested wetlands can shift the community composition to more flood tolerant tree species.

Increased ponding also creates favorable habitats suitable for nonnative and predatory fish, which can impact native amphibian populations (USGS, 2004; Delis *et al.*, 1996). In a study of Ontario wetlands, Hecnar and M'Closkey (1998) found a correlation between amphibian species richness and increased water depth. Other studies have determined that increased ponding resulted in a decrease in sensitive amphibian species and an increase in tolerant species such as the bullfrog, *Rana catesbeiana,* shown in Figure 11 (Delis e*t al.*, 1996; Rubbo and Kiesecker, 2005). In some cases, the most urbanized wetlands were found to contain only one species, the bullfrog (Reinelt *et al*. 1998).

Stormwater runoff can cause deeper ponding in wetlands that cannot rapidly discharge flows. This leads to changes in wetland type, function and quality.

Table 7. Recent Research on the Effects of Increased Ponding		
Key Finding(s)	**Location**	**Reference**
The functional responses of impacted forested wetlands were compared with non-impacted "reference" wetlands to determine the effects of highway construction. Water surface elevation, water depth, and tree stand density and mortality were indicative of changes in wetland functional change due to the presence of highways.	NC	Richardson and Nunnery, 1998
Excavated wetlands had deeper water levels (increased volume) and longer hydroperiods than natural wetlands. Water depth changes led to a change in plant community composition in disturbed wetlands. 90% of the experimental shallow-marsh plot areas of excavated wetlands lacked vegetation throughout the wetland	ND	Euliss and Mushet, 2004
Changes in land use resulted in increased ponding, which produced large scale changes in wetland vegetation.	WI	Owen, 1999
Changes in wetland hydrology affected plant communities (increase in exotic grasses) and species richness (50% decline over a few decades) within a wetland.	Ontario	Chow-Fraser *et al.*, 1998
Declines in plant species richness were observed in Puget Sound wetlands when water ponding depth exceeded two feet.	WA	Azous *et al.*, 1997
Invasive wetland plants outgrew perennials in four differing hydrologic regimes. Species sensitive to flooding were found in drier, groundwater-fed, and nutrient-poor environments. Noninvasive plants tolerated flooding but produced less biomass and were shorter lived than the invasive plants.	WI	Kercher and Zedler, 2004

Figure 11. Pollutant-tolerant bullfrog, *Rana catesbeiana*
Photograph by Jason Tomlinson

Increased Water Level Fluctuation

Water level fluctuation (WLF) is defined as the difference in the minimum and maximum water levels in a wetland for a given period of time and is often used to quantify a wetland's hydroperiod. The difference between ponding and water level fluctuation is the duration of time in which water levels remain elevated. In the case of ponding, the high water elevations are maintained for several seasons or even the entire year, whereas water level fluctuations are a temporary event measured in days or weeks. High water levels occur in response to moderate and large storms, but quickly return to a base level. Some researchers refer to WLF as the "bounce" in water levels during and after a storm event.

Water levels in most wetlands are dynamic and change on a seasonal or annual basis. Water levels tend to vary seasonally in wetlands that rely on local precipitation, while wetlands that rely more on groundwater tend to have more stable water levels (Mitsch and Gosselink, 1993). In urban watersheds, excessive stormwater runoff can sharply increase the amplitude of WLF for many wetland types.

The link between development in the CDA and increased WLF was first reported for palustrine wetlands in the Puget Sound lowlands of western Washington. Several researchers has confirmed a strong relationship between greater mean WLF and percent impervious cover (IC) in the CDA (Table 8). Taylor *et al*. (1995) found that when IC in the CDA exceeded about 4%, significant increases in both mean and maximum WLF could be detected in wetlands and determined that IC was the most important predictor of WLF.

Table 8. Recent Research on the Effects of Changes in Water Level Fluctuation

Key Finding(s)	Location	Reference
In a study of the impacts of urbanization on Puget sound wetlands, 78% of watersheds with >21% impervious area had wetland WLF ranges greater than 33.4 inches. Watersheds with more than 21% total imperviousness were more likely to have mean annual water level fluctuations greater than 7.9 inches. The relative frequency of thin-stemmed emergent plants in the wetlands decreased over time due to increasing WLF and mean depth. These species are important for amphibian breeding and egg attachment sites.	WA	Chin, 1996
Watersheds with < 5.5% IC had a mean wetland WLF < 8.3 inches. With IC > 21%, WLF exceeded 8.3 inches 89% of the time. Declines in plant species richness in emergent and scrub/shrub wetland zones occurred as WLF increased - in both zones where WLF was greater than 8.7 inches, richness decreased significantly.	WA	Horner *et al.*, 1997a
Significant increases in WLF were noted for wetlands draining the most developed watersheds.	WA	Reinelt and Horner, 1991
In Puget Sound wetlands, species richness was found to be significantly lower when water depths were more than two feet. Emergent plant zones with WLF >9.4 inches had fewer plant species than in zones with <9.4 inches. Wetlands with higher WLF had lower species richness and increasing dominance by invasive plants. Urbanization causes emergent meadows to become dominated by cattail or reed canary grass.	WA	Azous *et al.*, 1997

Table 8. Recent Research on the Effects of Changes in Water Level Fluctuation		
Key Finding(s)	**Location**	**Reference**
20% impervious cover from upstream development increased peak and volume of stormwater runoff to the point that it began to dominate the hydroperiod of downstream wetlands. Watershed forest cover, watershed IC, constriction of wetland outlet and ratio of wetland to watershed area had strongest influence on WLF.	WA	Reinelt and Taylor, 2001
From 1988-1995, 19 Puget Sound wetlands showed changes in wetland hydrology resulting from urbanization. In both scrub/shrub and emergent wetlands, plant richness was significantly negatively correlated with percentage impervious area within the watershed and mean WLF.	WA	Reinelt *et al.*, 1998
In the same study of 19 Puget Sound lowland wetlands (Reinelt *et al*, 1998), the ratio of wetland area to watershed area had a strong influence on WLF. As impervious area exceeded certain thresholds (3.5% and 20%), mean and maximum WLF were significantly higher in wetlands, respectively.	WA	Taylor *et al.*, 1995
Depth, duration, and frequency of inundation negatively influenced wetland plant community composition but depth was least important. Species and biomass differed by WLF. Sites that never flooded had the greatest biomass and richness, while those that continuously flooded had the least.	Australia	Cassanova and Brock, 2000
Large scale vegetation changes, such as increased dominance by invasive plants and fewer native species, occurred in emergent zones of palustrine wetlands receiving urban runoff, caused by WLF and other hydrologic changes.	WI	Owen, 1999
The growth of reed canary grass and native grass species were compared under four hydroperiods at two water depths. Reed canary grass was dominant due to high ratio of shoot length to biomass and its adaptable morphology.	WI	Ellison and Bedford, 1995
In a model, seed germination of annuals declined as water levels increased, but perennials germinated best in moderately flooded to drawdown conditions. The results explain the trend in increasing *Typha latifolia* (invasive) dominance in a Wisconsin wetland subject to water level increases.	WI	Ellison and Bedford, 1995
Changes in wetland hydrology affected plant communities (increase in exotic grasses) and species richness (50% decline over a few decades) within a wetland.	Ontario	Chow-Fraser *et al.*, 1998

A second distinct threshold in WLF amplitude has been observed for wetlands with more than 20% IC in their CDA (Chin, 1996 and Horner *et al.*, 1997a). The mean WLF for wetlands in this group was about eight inches and appeared to have a major impact on wetland biota. Subsequent work by Reinelt and Taylor (2001) confirmed that increased stormwater was the primary factor dominating the hydroperiods of urban wetlands that had CDAs with more than 20% IC.

While IC was found by most researchers to be the most important variable to predict WLF for urban wetlands, forest cover and outlet constriction were also useful (Reinelt *et al*., 1998). Taylor *et al*. (1995) reported that the ratio of wetland area to the CDA area had a strong influence. The same basic relationship between urbanization and increasing wetland WLF has been reported for wetlands in Australia and Wisconsin (Cassanova and Brock, 2000 and Owen, 1999).

The impact of increased WLF on urban wetland function and quality has been extensively studied (summarized in Table 8). The response of the plant and animal communities within the wetland to increased WLF is a consistent decline in diversity and often an increase in invasive species (e.g., Cooke and Azous, 1993; Ehrenfeld and Schneider, 1993; Owen, 1999).

Most wetland plant species are closely adapted to specific wetland hydroperiods and are not very tolerant of major changes in WLF. Thus, most wetland plant communities develop in response to a fairly narrow WLF range. As WLF increases, plant communities respond in a predictable manner—a loss in species richness, a loss of sensitive species, and an increase in invasive plant coverage (see review in Table 8). Invasive species tend to dominate the wetland community as they are often more tolerant of hydrologic change than native species.

Considerable evidence demonstrates that wetland communities are particularly vulnerable to increases in water level fluctuation (WLF) caused by excessive stormwater runoff and that WLF provides more favorable conditions for the spread of invasive plant species.

Plant richness declined sharply when WLF exceeded nine to 10 inches in Puget Sound wetlands, especially during the growing season (Azous *et al.*, 1997; Horner *et al*., 1997a). Plant richness was also diminished in wetlands that experienced WLF fluctuations more than three times per month (Azous *et al*., 1997). Invasive species, on the other hand, appear to tolerate and even thrive under high WLF conditions. Several studies have noted that invasive plant species such as *Phalaris* and *Typha* grow best in wetlands with variable WLF generated by stormwater runoff (Azous and Horner, 1997; Owen, 1999; Kercher and Zedler, 2004; Mahaney *et al*., 2004; Miller and Zedler, 2003).

In Washington wetlands, invasive reed canary grass grew best in drier areas with very high seasonal WLF, whereas cattail and soft rush, *Juncus effusus*, were found in areas which had WLF during the growing season (Cooke and Azous, 1997; see Figure 12). Likewise, the combination of WLF and runoff resulted in invasive *Phalaris* dominance and the loss of native species in wetlands receiving stormwater runoff (Miller and Zedler, 2003). For more information, Zedler and Kercher (2004) provide an in-depth review of the causes and consequences of invasive plants in wetlands.

Water level fluctuation appears to be a primary factor influencing amphibian populations in urban wetlands (Figure 13). For example, Chin (1996) observed that wetlands with an annual WLF greater than 8.7 inches and impervious cover greater than 21% had fewer than three amphibian species. Richter and Azous (1995) found that wetlands with WLF less than eight inches had higher amphibian species richness (average of 5 species). Declines in amphibian species richness may be caused by diminished reproductive success (Chin, 1996). For example, Richter and Azous (1995) determined that declining water levels can strand larvae or expose amphibian eggs attached to emergent vegetation.

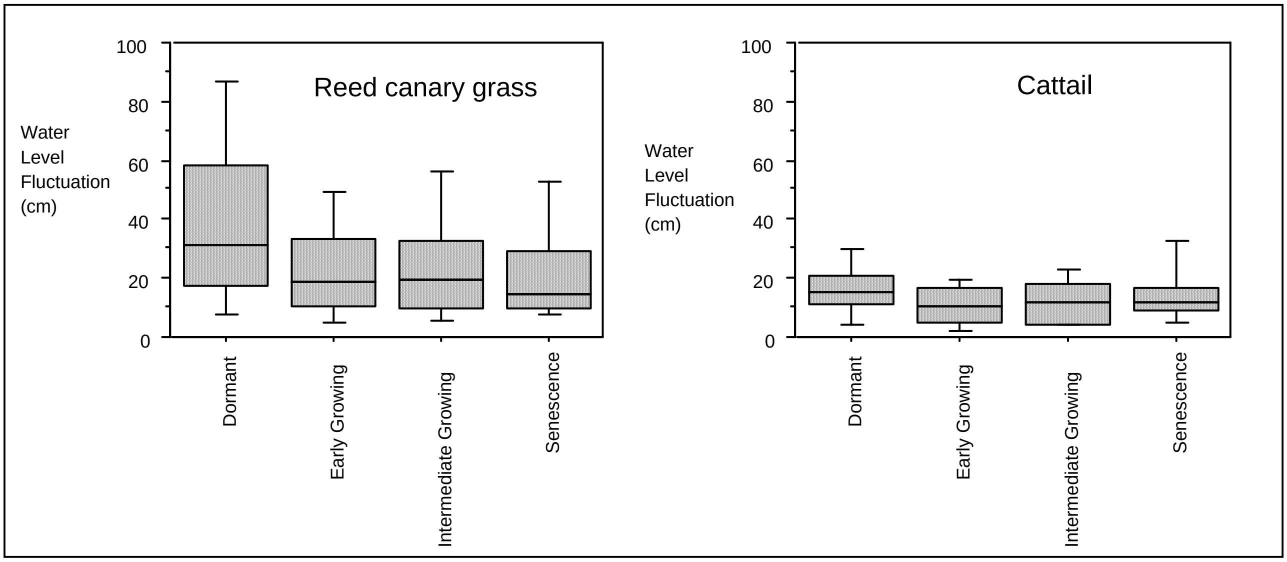

Figure 12. Hydrology and water level fluctuation of reed canary grass and cattail
Source: Cooke and Azous, 1997

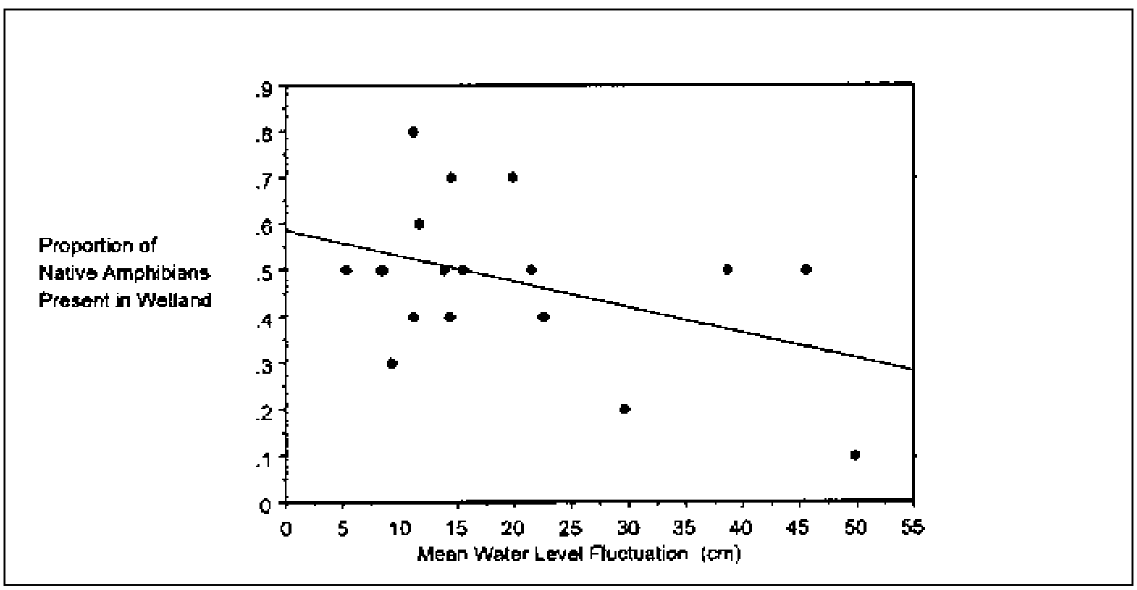

Figure 13: Relationship between amphibian species presence and water level fluctuations (Source: Reinelt *et al*., 1998b; used with kind permission from Springer Science and Business Media)

Flow Constrictions

The impact of flow constrictions on wetlands has been documented by several researchers. The hydrologic changes caused by the constriction may involve increased ponding, greater water level fluctuation, or hydrologic drought, depending on whether the wetland is located above or below the crossing. For this reason, the literature on the impact of crossing-related flow constrictions on wetlands is discussed separately and is summarized in Table 9.

Researchers have shown that crossings can exert a localized impact on wetland quality that extends in an upstream and downstream direction. The impact to plant and animal communities appears to be greatest for freshwater wetlands above the crossing due to increased ponding and WLF.

Richardson and Nunnery (1998) found that wetlands upstream from highway culverts had higher water surface elevations, greater surface area, and greater ponding depths compared to downstream reference wetlands. These hydrologic changes, in turn, caused increased tree mortality and sediment accumulation, and reduced availability of soil nutrients. Richardson and Nunnery (2001) compared wetland function at the upstream and downstream wetland sites, and found upstream wetlands experienced a 68% reduction in functional quality. Levine *et al*. (2003) found that pool habitat in stream sections above and below channel constrictions tends to decrease, which may alter the water table elevation and hydrology of adjacent wetlands.

Flow constrictions caused by road crossings have also been shown to alter aquatic invertebrate community structure in urban wetlands (King *et al*., 2000). In a study of highway crossings at forested wetlands in North Carolina, King reported a surprising increase in aquatic invertebrate species richness and percent herbivores within 125 feet of highway crossings (King *et al*., 2000). These changes were attributed to a loss of forest canopy at the road crossing, which increased sunlight penetration, shifted primary production from trees to macrophytes and algae, and increased the number of herbivorous taxa present. Other aquatic invertebrate species responded negatively to crossings, such as the damselfly (*Ischnura* sp.) and fingernail clam (*Sphaerium* sp.). Wetland aquatic invertebrate communities appear to be closely related to habitat factors produced by the crossing such as water quality, forest cover, and/or flow levels.

Crossings of tidal wetlands were observed to increase sediment deposition and nutrient loadings. Reduced salinity in these tidal wetlands shifted populations of aquatic invertebrates and fish and led to a barrier for fish moving upstream to spawn (Richardson *et al*., 2003). Sturdevant *et al.* (2002) studied an urban salt marsh adjacent to the New York/New Jersey Harbor that had once been tidally inundated, but was subsequently ditched and impounded. The ecological function of the impounded tidal marsh was greatly reduced in comparison to three adjacent tidal marshes that were not impounded. *Phragmites australis* dominated the impounded marsh, while two of the three unimpounded marshes were dominated by native *Spartina* grasses.

Table 9. Recent Research on the Indirect Physical Impacts of Road Crossings on Wetlands		
Key Finding(s)	**Location**	**Reference**
Findings include decreased pool habitat 165-330 feet upstream and downstream of bridges; road crossings caused changes in bank stability upstream and downstream.	NC	Levine *et al*., 2003
The functional responses of impacted forested wetlands were compared with a non-impacted "reference" wetland to determine the effects of highway construction. Water surface elevation, water depth, and tree stand density and mortality were indicative of declines in wetland function due to the presence of highways.	NC	Richardson and Nunnery, 1998
Research showed a 68% decline in ecosystem function of a disturbed wetland upstream of a highway crossing.	NC	Richardson and Nunnery, 2001
Temporary culverts appeared to increase runoff, impede fluxes of water from floods and tides, and change soil surface elevations. These impacts may be temporary, but further research is needed.	NC	Richardson, Flanagan, and King, 2003
The effects of dams and road construction included modified water flow and increased variability of hydrologic patterns (WLF) in Atlantic cedar wetlands.	NJ	Ehrenfeld and Schneider, 1990
Nine wetlands that are crossed by a highway showed signs of wetland drainage, and the wetlands are smaller than before construction. Wetland drainage effects extended outward from the road for distances varying from 54.6 to 546 yards. Five wetlands were affected at about 328 feet from road.	MA	Forman and Deblinger, 2000
In forested wetlands, highway crossing impacts were likely highly correlated with change in forest canopy coverage. Significant decrease in crown closure and basal area were found in proximity to the highway.	NC	King *et al*., 2000
Plant species richness decreased relative to increases in paved road density at all distances studied with the most impact on species richness found within 0.6 miles of the road.	Ontario	Findlay and Houlahan, 1997

<u>Decreased Groundwater Discharge</u>

There are virtually no research studies that have examined the link between the diminished groundwater recharge in the CDA and wetland quality. The lack of research is not surprising given how challenging it is to monitor the movement of groundwater into and out of wetlands. One study by Ehrenfeld and Schneider (1990) noted changes in the water table at wetlands with residential land use and found some evidence that they had been altered as a result of adjacent development.

While the link between decreased groundwater recharge in the CDA and diminished wetland quality seems tempting, there is simply not enough scientific evidence to determine whether it actually exists.

A greater number of studies have documented how decreased groundwater recharge influences dry weather flow in small headwater streams. The basic process is that urbanization reduces the volume of groundwater available to sustain baseflow in small streams. Indeed, several studies have shown a decrease in dry weather stream flows in response to urbanization in humid watersheds (Klein, 1979; Saravanapavan, 2002; Simmons and Reynolds, 1982;), although the opposite may occur in streams in arid and semi-arid climates due to water from irrigation nuisance flows. Therefore, it is conceivable that diminished groundwater recharge in the CDA can influence the water budget of riparian wetlands that depend on streams as their primary source of water. The effect of diminished recharge may also be significant for other wetland types whose water budget is dominated by groundwater. In any event, additional research is needed to define the extent to which development in the CDA diminishes groundwater discharge to wetlands, and what the expected hydrologic and biological response of the wetland will be.

Hydrologic Drought in Riparian Wetlands

Development in the CDA of headwater streams has been strongly linked to active channel enlargement by widening of the stream banks or lowering of the streambed (Schueler, 2001b). These urban stream channels may incise over time following the general process depicted in Figure 14. As the channel deepens, the local water table drops, often to the point where it is below the rooting depth of riparian forests and plants (Schueler and Brown, 2004). A second consequence of stream incision is that channels deepen and enlarge such that riparian wetlands become disconnected from the stream. The floodwaters that once spilled over the banks to supply water to riparian wetlands are now confined within the deeper and enlarged stream channel. Riparian wetlands that depend on occasional flooding and baseflow to sustain their hydroperiod can face a condition termed hydrologic drought as urbanization increases in the CDA. Hydrologic drought occurs when a riparian wetland does not receive adequate water to sustain its hydric soils and vegetation.

The influence of urban stream channel incision, dropping water tables and floodplain disconnection may well play a negative role in riparian wetlands, but much more research is needed to define over which stream sizes and floodplain widths it can occur.

Although most urban stream geomorphologists recognize how urban stream channel incision disconnects floodplains and drops water tables, researchers have yet to examine how these changes influence riparian wetland quality. At this point, the only research study on hydrologic drought in urban riparian wetlands is by Groffman *et al* (2003) and references cited therein. Groffman *et al.* (2003) suggests that urban stream incision, declining water tables, and floodplain disconnection have secondary effects on the soils and plants of riparian wetlands. Although considerable anecdotal evidence exists for the concept of hydrologic drought, it is not clear over which stream orders it is most pronounced and how far laterally it extends into the floodplain. Clearly, greater collaboration between wetland scientists and urban stream geomorphologists is needed to understand the wetland dynamics associated with hydrologic drought.

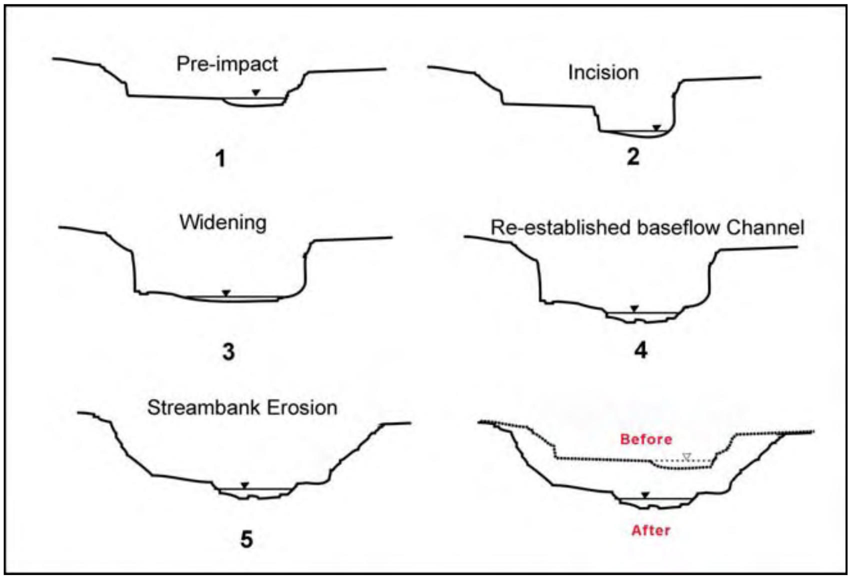

Figure 14. Progressive Stages of Channel Incision
Source: Schumm,1999

Water Quality Stressors
While wetlands can provide some pollutant removal function, wetland communities can also be stressed when pollutant loads are excessive. This section reviews what is known about water quality stressors to wetlands, with an emphasis on wetland types that are most sensitive to changes in water quality. The four major water quality stressors described in this section include:

1. Sediment Deposition
2. Pollutant Accumulation in Wetland Sediments
3. Nutrient Enrichment
4. Chloride Discharges

Each of the four water quality stressors can produce significant changes in the functions and quality of urban wetlands, including excessive primary productivity, spread of invasive species, and loss of sensitive plant and animal species. This section reviews research on each stressor in a progressive manner. It begins by quantifying how pollutant loads increase as a result of land development in the contributing drainage areas. Next, research is profiled to show how pollutants alter water quality within urban wetlands. Lastly, research on how plants and animals respond to declining water quality in urban wetlands is described. In some cases, our understanding of wetland water quality stressors can be inferred from research conducted on created wetlands specifically designed for stormwater treatment.

<u>Sediment Deposition</u>
Sediment is generated from three major sources in the CDA: stormwater runoff from impervious surfaces, erosion from exposed construction sites, and upstream channel erosion. Construction sites can be a significant source of sediment to a wetland, particularly if adequate erosion and sediment control practices are not installed. Sediment loads from uncontrolled construction sites

are estimated at 100 tons per acre per year, while sediment loads from cropland with poor conservation practices can reach 20 tons per acre per year. This is as much as 500 times more sediment than from a forest with deep soils, where sediment loads range from 200 to 400 pounds per acre per year (Envirocast 2003).

Most urban wetlands receive greater sediment loads than wetlands located in undisturbed settings.

Schueler and Lugbill (1990) studied sediment concentrations from construction sites in Maryland and found that uncontrolled sites can deliver seven times more sediment than controlled sites. Other studies show sediment concentrations from uncontrolled construction sites as high as 7,363 mg/l (Horner *et al*., 1990). Streambank erosion can often be the dominant sediment source to downstream wetlands, when increased runoff in the CDA creates higher stream flows that erode stream channels. In fact, urban streambank erosion can comprise as much as 75% of the sediment budget of urban streams (Dartiguenave *et al*., 1997; Trimble, 1997; CWP, 2003).

As land is developed in the CDA, sediment loadings to downgradient wetlands increase. Even if soils and streambanks in the CDA are stable, urban wetlands can expect to receive greater sediment loads than non-urban wetlands. Reinelt and Horner (1991) found that suspended sediment concentrations were significantly higher in urban wetlands compared to non-urban ones.

Relatively few researchers have measured sediment deposition rates within urban wetlands. One study by Bazemore *et al.* (1991) evaluated sediment deposition within forested wetlands near highway crossings and found greater deposition rates at wetlands with longer hydroperiods. While no significant overall increase in sediment deposition rates at highway crossings was observed, greater sediment deposition rates were noted upstream and downstream of constricted crossings. King *et al*. (2000) investigated sediment deposition rates at road crossings in forested wetlands and found that deposition within 33 feet of highway crossing was significantly higher than in reference wetlands more than 130 feet distant from the crossing.

A much larger number of studies have tested the impact of sediment deposition on wetland plants in field and greenhouse experiments. The research generally indicates that sediment deposition can harm the wetland plant community by reducing germination of wetland plant seeds, reducing growth and survival of native species, and favoring conditions for invasive wetland plants. A summary of research of the impact of sediment deposition on wetland plant communities can be found in Table 10.

When sediments from the CDA are deposited in urban wetlands, the structure of the plant community shifts away from sensitive species and towards invasive species.

Werner and Zedler (2002) correlated higher sediment deposition rates to a decrease in native sedge meadow plant species, and reported that an average of 1.2 native species were lost for every 4 inches of sediment deposited. The authors concluded that sediment deposition reduces microtopographic variation and surface area in the wetland that was critical for the growth of native species. At the same time, higher sediment deposition rates favored invasive species tolerant of sediment such as reed canary grass (Werner and Zedler, 2002).

Increased sediment transport to wetlands can cause alterations in plant community dynamics. Wardrop and Brooks (1998) found that sedimentation influences the ability of seeds to germinate and grow by altering light availability, temperature, and oxygen levels in the soil. Another study of three common wetland species showed sediment deposition generally reduced seedling establishment and was responsible for a decline in species richness and density (Mahaney *et al*., 2004). Germination experiments by Gleason *et al*. (2003) and Wardrop and Brooks (1998) found that as little as 0.2 inches of sediment deposition was enough to reduce wetland seedling emergence, although plant response varied by species.

Sediment deposition can alter the diversity of the wetland aquatic invertebrate community. For example, Martin and Neely (2001) found that aquatic invertebrate density was lower in wetland plots that received higher sediment loads than those that received less sediment. Species that were adversely affected by sediment include *Coleoptera* and *Diptera* larvae, *Megaloptera, Odonata*, *Gastropoda*, and *Spaeriidae* clams. The scraper functional feeding group displayed the most adverse response to sediment deposition, likely due to the loss of the periphyton on which they feed (Martin and Neely, 2001). Similarly, experiments in North Dakota indicated that even sediment deposition as little as 0.2 inches caused a 99.7% reduction in invertebrate emergence from wetland soils (Gleason *et al*., 2003).

Table 10. Recent Studies on the Impacts of Sediment Accumulation on Wetlands

Key Finding(s)	Location	Reference
Three different wetland types were simulated in a greenhouse. Sediment lowered community biomass, diversity, and richness and reduced seedling establishment.	PA	Mahaney *et al.*, 2004
Sedimentation was measured in 25 wetlands representing a variety of HGM subclasses. Sedimentation rates differed by HGM class and were highest in headwater floodplains, riparian depressions, impoundments, and slopes. Plant species that were intolerant of sediment deposition had a significant decrease in germination with as little as 0.2 inches of sediment accumulation.	PA	Wardrop and Brooks, 1998
Different sediment depths (0 to 0.8 inches) caused various impacts on plants. The minimum experimental depth of 0.5 cm of sediment reduced the total seedling emergence by 91.7% and caused a reduction of 99.7 in total invertebrate emergence.	MT, ND, SD, MN	Gleason *et al.,* 2003
In sedge meadows, increased sediment deposition resulted in a decrease in native species richness and increase in invasive reed canary grass. An estimated 1.2 species were lost for every four inches of sediment deposition in sedge meadows.	WI	Werner and Zedler, 2002
In forested wetlands, sedimentation rates were highest immediately adjacent to highway crossings (33 feet) but similar to the reference wetland beyond 130 feet.	NC	King *et al.*, 2000
Greenhouse experiments showed 0.8 inches of sediment significantly increased aboveground biomass of broadleaf arrowhead (*Sagittaria latifolia*), while field plots showed no differences in plant diversity, biomass, or stem density. Paired field and greenhouse studies showed increases in bulk density, which would alter hydrologic functions of wetland if these sediment depths occurred over a larger area.	NH	Koning, 2004

Pollutant Accumulation in Wetland Sediments

Urban stormwater runoff carries with it many different pollutants, including hydrocarbons and metals such as cadmium, copper, lead, nickel, and zinc. Research has shown that vehicle emissions are a dominant source for many metals of concern (EOA, Inc., 2001), although atmospheric deposition, roof surfaces, and snowmelt may also be important sources. Table 11 presents median national Event Mean Concentrations (EMC) for trace metals found in urban stormwater runoff from different land uses as reported in the most recent National Stormwater Quality Database (Pitt *et al*., 2004).

The top sediments of urban wetlands receiving stormwater inputs are enriched with a distinct signature of elevated trace metals and hydrocarbons.

The term hydrocarbons refers to a large group of organic chemicals present in petroleum products that are produced during combustion (Note: In this discussion, the term Polycyclic Aromatic Hydrocarbons, or PAH is used interchangeably with hydrocarbons although they are not exactly the same). Hydrocarbons tend to persist in the environment and may accumulate in sediment organisms and shellfish, yet toxicity levels are not well established. Hydrocarbons concentrations are significantly greater in runoff from industrial areas, gas stations, and roads; recent studies in Texas indicate that PAH concentrations in runoff from coal-tar sealed parking lots are significantly higher than other types of urban land cover (Schueler and Shepp, 1992; Mahler *et al*., 2005). Therefore, wetlands with these uses in their CDAs may be at risk. Typical concentrations of hydrocarbons found in stormwater runoff are presented in Table 12.

Table 11. Median National Event Mean Concentrations of Five Trace Metals in Stormwater Runoff for Different Land Uses

Parameter (µg/L)	Residential	Commercial	Industrial	Freeways	Open Space
Total Cadmium	0.5	0.9	2	1	0.5
Total Copper	12	17	22	35	5.3
Total Lead	12	18	25	25	5
Total Nickel	5.4	7	16	9	ND
Total Zinc	73	150	210	200	39

ND = not detected, or insufficient data to present as a median value.
Source: Pitt *et al.*, 2004

Table 12. Hydrocarbon EMCs in Urban Stormwater Runoff

Hydrocarbon Indicator	EMC (land use)	Number of Events	Location	Source
PAH (µg/l)	3.2*	12	MA	Menzie-Cura, 1995
	7.1	19	MA	Menzie-Cura, 1995
	13.4	N/R	WI	Crunkilton *et al*., 1995
Oil and Grease (mg/l)	1.7**, 9 (C), 3 (I)	30	TX	Baird *et al*., 1996
	3	N/R	U.S.	USEPA, 1983
	5.4*	8	MA	Menzie-Cura, 1995

Table 12. Hydrocarbon EMCs in Urban Stormwater Runoff				
Hydrocarbon Indicator	**EMC (land use)**	**Number of Events**	**Location**	**Source**
	3.5	10	MA	Menzie-Cura, 1995
	3.89 (R), 13.13 (C), 7.10 (I)	N/R	CA	Silverman *et al*., 1988
	2.35 (R), 5.63 (C), 4.86 (I)	107	MD	Barr, 1997
* Geometric mean N/R - Not Reported **Median R - Residential, C - Commercial, I - Industrial				

When metals and hydrocarbons in stormwater runoff from the CDA enter wetlands, they eventually accumulate in wetland sediments. Pollutants trapped in wetland sediments can re-enter the water phase or migrate downward, and in some cases, into the groundwater. The primary concern with metals and hydrocarbons are their potential toxicity and bioaccumulation in aquatic organisms and plants. For a complete review of wetland sediment toxicity, consult Burton *et al*. (1992) or Baudo *et al*. (1990).

There is growing evidence that both metals and hydrocarbons are accumulating in urban wetland sediments due to stormwater pollution. For example, in a review of sediment chemistry in more than 50 stormwater wetlands and ponds, trace metal concentrations were five to 30 times higher in the muck layer than underlying soils and had a distinct signature and vertical distribution (Schueler, 2000a). A similar pattern in metal distribution was found over time in a natural urban forested wetland that received urban stormwater runoff (Sanders, 2002). Significant increases in cadmium, copper, zinc, and nickel were reported in sediments deposited over a 30-year period (Table 13). Gernes and Helgen (2002) found higher concentrations of copper, zinc, and lead in natural depressional wetlands with increased urbanization as compared to a reference natural depressional wetland. Horner *et al.* (1997b) also reported that sediments in an urban stormwater wetland in Puget Sound region had copper, zinc, and arsenic levels that were up to four times greater than non-urban wetlands. This suggests that natural wetlands that receive stormwater runoff may also receive similar elevated levels of pollutants.

Table 13. Comparison of Historic and Current Mean Metal Concentrations in the Sediment of an Urban Forested Wetland (Sanders, 2002)		
Metal	**Pre-1963 sediment**	**Post-1963 sediment**
Zn	56.2	95*
Pb	32	45
Cr	11	10.7
Ni	11.1	17*
Cu	5.32	22.7*
Cd	0.32	2.0*
*Mean is significantly different from pre-1963 sediment		

Only a handful of studies have explored hydrocarbon accumulation in the sediments of urban wetlands, but the few reports indicate urban wetlands have elevated levels (Seattle Metro, 1993 and Watts, 2006). Bryan and Langston (1992) reported that hydrocarbon concentration in urban wetland sediments were frequently three to five orders of magnitude greater than those found in the water column. Paul *et al*., (2002) found that sediment contamination in small estuaries increased with increasing percentage of urban land use and decreasing area of non-forested wetlands.

Research on the possible biological impact of metal and hydrocarbon accumulation in wetland sediments is inconclusive, and is summarized in Table 14 below. While metal concentrations in stormwater are not usually high enough to cause acute toxicity to aquatic organisms, the concentration of metals in bottom sediments are of greater concern (Field and Pitt, 1990). Pollutants trapped in sediment may re-enter the food webs either through uptake by plants or aquatic organisms. The primary concern is the potential for bioaccumulation and toxicity to aquatic organisms.

Table 14. Review of Contaminants in Wetland Sediments

Parameter	Key Finding(s)	Location	Reference
Metals	In leaf analysis of plants receiving stormwater runoff, concentrations of Zn and Cu were higher than Pb, and Cd.	WA	Cooke and Azous, 1993
	Sewage sludge was applied to a freshwater tidal wetland to examine retention of heavy metals. Results showed that soil in treatment areas retained significantly higher levels of all metals than the control.	NJ	Dubinski *et al.*, 1986
	Natural urban depressional wetlands had considerably higher concentrations of heavy metals (Cu, Pb, Zn) than non-urbanized, natural, depressional, reference wetlands.	MN	Gernes and Helgen, 2002
	Urbanized wetlands had higher sediment levels (up to four times greater) of copper, zinc, arsenic compared with those in non-urban wetlands.	WA	Horner *et al.*, 1997b
Metals, cont.	Two wetlands in Florida were compared to assess the effect of highway runoff. One wetland had pre-treatment of runoff before discharging it to the wetland, while the second wetland received untreated stormwater. Metal concentrations in sediments in the wetland receiving untreated runoff were an order of magnitude higher than in the wetland receiving untreated stormwater.	FL	Schiffer, 1989
	A study of stormwater wetlands in Seattle found that Zn and Pb were higher in the roots than in the emergent vegetation	WA	Seattle Metro, 1993
PAHs and oil/grease	PAH and oil and grease concentrations in stormwater ponds exceeded Ontario standards.	Ontario	Bishop *et al.*, 1999

Trace metals and hydrocarbons are clearly accumulating in the tissues of plants and animals in urban wetlands exposed to stormwater, although it is unclear whether the reported levels are causing toxicity in the food chain.

Research on urban wetland plants indicates that a larger proportion of metals are stored in the roots. A study of stormwater wetlands in Seattle found that zinc and lead were higher in the roots than in the emergent vegetation except in burreed (*Sparganium sp*.; Figure 15; Seattle Metro, 1993). In general, metal uptake and allocation is species specific. The bulk of the contaminants are stored in the roots, not the stem or leaves, but there are exceptions (Lepp, 1981; Dunbabin and Bowmer, 1992).

As metals accumulate in plant and animal tissue, they may have the potential to cause toxicity. Although extensive literature exists on metal toxicity for estuarine sediments, very little research is available on sediment metal toxicity for freshwater wetlands exposed to stormwater runoff. In general, symptoms of metal toxicity include vulnerability to disease, stunted growth, and alterations of the food web for bottom dwelling organisms. Because metals are so concentrated in sediments that the bioavailability of even small amounts of the total sediment metal is highly important for bottom dwelling organisms (Bryan and Langston, 1992).

Predicting toxicity of metals in sediment is difficult due to a host of processes that control bioavailability and fate. Temperature, pH, and salinity are just a few of the factors that can impact metal toxicity and availability (Resh and Rosenberg, 1984; Cherry *et al*., 2001; Tomson *et al.*, 2003; Du Laing *et al.*, 2002). Lastly, the amount of metals in sediment depends on the size of the sediment, as more metals will accumulate on fine sediment (Gibbs, 1973). For a complete discussion of metal bioavailability, see John and Leventhal (1995).

Plants absorb hydrocarbons from bottom sediment and readily move them to above-ground tissue, although hydrocarbons, like metals, tend to be found in higher concentrations in wetland plant roots than in the leaves (Seattle Metro, 1993 and Watts *et al.*, 2006). Watts *et al.* (2006) reported that hydrocarbon levels in the roots of *Spartina alterniflora* were strongly correlated with PAH concentrations in contaminated sediment, although hydrocarbon levels were much lower in the roots than in the sediment. Bioaccumulation as a result of wetland herbivory does not appear to be a significant route of hydrocarbon exposure, and biomagnification does not occur for terrestrial species (USEPA, 2003). Still, researchers have documented elevated PAH levels in dragonflies, crayfish, clams, and fish (Masterson and Bannerman,1994; Moring and Rose 1997; and Velinsky and Cummins 1994). Possible effects of hydrocarbon toxicity on aquatic organisms include reduced diversity, inhibited reproduction, delayed emergence, sediment avoidance, and mortality (USEPA, 2003). Culbertson *et al*. (2005) found that more than 30 years after an oil spill, hydrocarbon contamination still existed to depths of 10-14 inches in salt marsh sediments of Massachusetts. While vegetation appeared to have recovered from the spill, fiddler crab populations, *Uca pugnax*, still showed measurable effects.

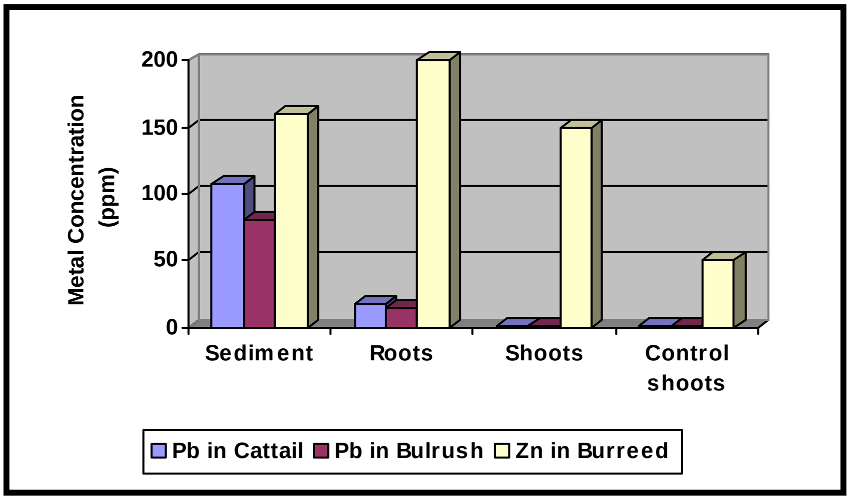

Figure 15. Pollutant levels found in sediment and in three species of wetland plants (Source: Seattle Metro, 1993)

Nutrient Enrichment

Nitrogen and phosphorus are essential nutrients in a wetland ecosystem, but when present in excess concentrations, they can become a stressor. The nutrient load generated by the CDA is influenced by many factors, but turf and impervious cover are perhaps the most important. Both turf and impervious cover generate higher nitrogen and phosphorus concentrations in stormwater runoff than forest cover, and together they comprise the majority of land cover created during land development (Table 15). In addition, turf and impervious cover generate more runoff during each storm event as compared to meadow or forest, so the total nutrient load discharged to a wetland from developed areas can increase by a factor of 5 to 20. Annual nutrient loading is the product of runoff volume and pollutant concentration yields the annual nutrient load. Figure 16 presents a comparison of the annual nutrient loading from forest, turf, and impervious cover; these loadings were calculated using the Simple Method (Schueler, 1987) and the runoff coefficients and nutrient concentrations shown in Table 6 and Table 15, assuming an average annual rainfall of 40 inches.

Urban wetlands exposed to stormwater runoff may receive nutrient loadings 5 to 20 times greater than undisturbed CDAs managed in a natural condition.

Table 15. Median Nutrient Concentrations in Stormwater			
Constituent	**Forest Cover [1]**	**Turf Cover [2]**	**Impervious Cover [3]**
Total Phosphorus	0.25 mg/l	1.9 mg/l	0.4 mg/l
Total Nitrogen	1.5 mg/l	9.7 mg/l	1.9 mg/l

1: From Mostaghimi, *et al*. (1994) and USGS (1999).
2: Grand mean of Garn (2002); Waschbusch, *et al*. (2000); Steuer, *et al*. (1997); and Bannerman, *et al*. (1993) turf runoff monitoring data.
3: Grand mean of all reported impervious cover source area monitoring data in CWP, 2003 (Table 19).

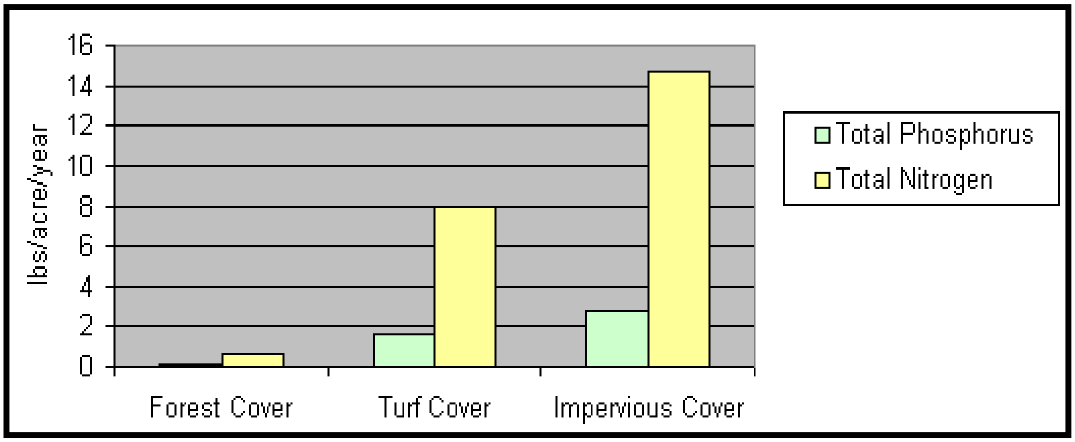

Figure 16. Nutrient loading from three different land covers

Brenner *et al*. (2000) used sediment cores to track nutrient accumulation rates in 11 Florida marshes since the early 20th century. Nitrogen accumulation rates in wetland sediment increased 1.6-3.7 fold, and phosphorus accumulation rates increased 2.3-17.0 fold since 1920. The sediment nutrient enrichment was attributed to agricultural and urban nonpoint source pollution. Houlahan and Findlay (2003) found a negative correlation between nitrogen levels in an Ontario wetland and the amount of adjacent forest cover present.

Table 16 summarizes some recent studies on the impact of nutrient enrichment on urban wetlands. Although only a few studies have addressed this topic, nutrient enrichment appears to be an important factor influencing the vegetation dynamics within wetlands (see U.S.EPA, 2002d). Wetlands enriched by nutrients often increase their plant productivity and may also shift their community structure – the most common scenario is that nutrient-sensitive species are replaced by species tolerant of high nutrient loading (USEPA, 2002d). Frequently these species can outgrow and out-compete native species.

Nutrient enrichment can alter the composition of urban wetland plant communities. For example, Woo and Zedler (2002) conducted greenhouse and field experiments to determine if nutrients could cause a sedge meadow to become dominated by an invasive species of cattails. They found that after one season with added fertilizer, the cattails more easily incorporated the excess nutrients into its tissues, outgrew native vegetation in density, height and biomass, and began to dominate a sedge meadow.

Miller and Zedler (2003) conducted experiments to determine the effects of flooding on the growth of native and invasive species. They concluded that water quality changes due to nutrient-rich stormwater runoff delivery were more likely the cause for the spread of the invasive *Phalaris arundinacea.* They also concluded that a reduction in stormwater volume could also reduce nutrient transport to wetlands, which may reduce the risk for remnant natural wetlands to become dominated by *Phalaris.*

Nutrient enrichment in urban wetlands increases overall productivity of the wetland but favors the spread of invasive wetland plants.

Gernes and Helgen (2002) reported that intolerant invertebrate taxa were absent from urban wetlands in Minnesota wetlands exposed to high levels of phosphorus and nitrogen (Figure 17). They reported that both invertebrate community index scores and the composite number of genera of *Ephemeroptera* (mayflies), *Trichoptera* (caddisflies), *Sphaeriidae* (fingernail clam), and *Odonata* (dragonflies) were negatively related to concentrations of phosphorus and nitrogen in wetlands.

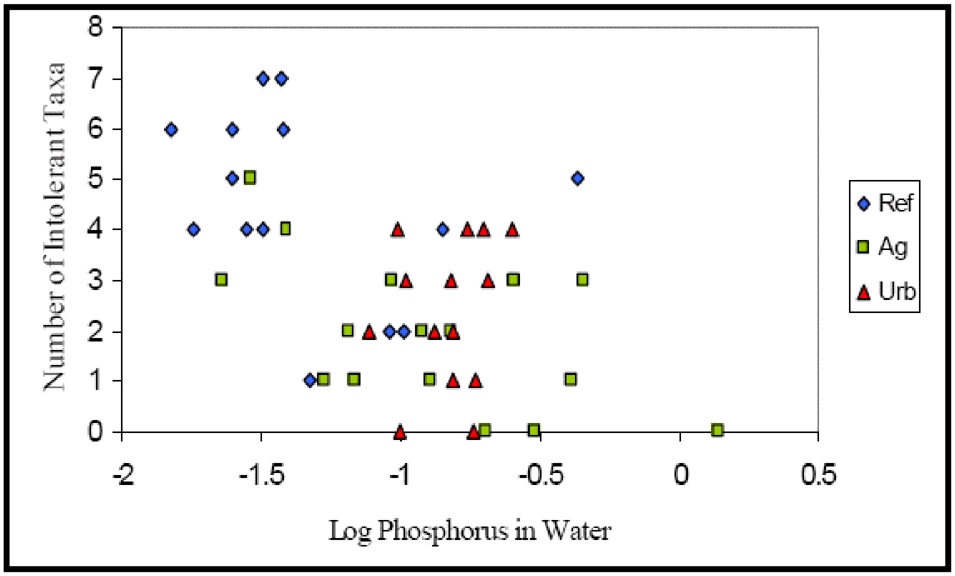

Figure 17. The effects of phosphorus on intolerant invertebrate taxa in Minnesota wetlands (from U.S. EPA, 2002a)

Table 16. Studies on Nutrient Enrichment of Wetlands		
Key Finding(s)	**Location**	**Reference**
Historical water quality changes in a Lake Ontario marsh were examined from 1973 to 1998. Changes in nutrient concentrations combined with increasing WLF have resulted in long-term changes in the planktonic and benthic communities.	Ontario	Chow-Fraser *et al.*, 1998
Within approximately 1.4 miles of wetland's edge, nutrient (phosphorus and nitrogen) levels were negatively correlated with adjacent forest cover. Nitrate was found to be positively correlated with road density with the most significant effect found at 1,640 feet from the road. They concluded that in order to sustain a high wetland quality, a regional landscape conservation approach is needed (rather than narrow buffers around wetlands).	Ontario	Houlahan and Findlay, 2003
Nutrients were added to field plots of a sedge meadow that partially surrounded a stand of cattails. The native sedge meadow grass did not have additional growth, whereas the invasive cattails increased in size suggesting that nutrients can cause a natural wetland to be invaded by invasive species.	WI	Woo and Zedler, 2002
Atlantic white cedar swamps were studied to examine water quality, hydrology and changes in plant species composition. Degradation of water quality resulted from urbanization but changes in water quality (NH_4 and PO_4) at any particular site were not predictable.	NJ	Ehrenfeld and Schneider, 1993
Long term ecological effects of low level phosphorus enrichment were studied in the FL Everglades over 5 years. Concentrations of P in periphyton & detritus were elevated by the first year; macrophytes by years 3 and 4. Water Total P enrichment was detected in year 5.	FL	Gaiser *et al.*, 2005

Chloride Discharges

Chloride can be a major wetland stressor in Northern latitudes due to the application of road salt in winter (Figure 18). Chloride is one of the main components of road salt, which is most often sodium chloride, but may also be blended with calcium or magnesium chloride. Nationally, road salt use ranges from 10 million to 20 million tons per year, with average annual application rates of up to 19.4 tons per lane mile (Salt Institute, 2001; TRB, 1991). Chloride in snowmelt runoff eventually makes its way into nearby streams, lakes, groundwater, and wetlands. Chloride is extremely soluble in water, so there is virtually no way to remove it once it gets into surface waters, and it can contaminate drinking water supplies (EC, 2001). Chloride moves freely through surface and groundwater, and its accumulation and persistence pose risks to wetlands.

Road salt accumulates in snowpacks and is ultimately released in snowmelt runoff in spring, although elevated levels of chloride can persist through summer (Demers and Sage, 1990). Typical event mean concentrations for chloride found in snowmelt and stormwater runoff are presented in Table 17.

Figure 18. Road salt is a major source of chloride to surface waters (Source: Beloit College)

Table 17. EMCs for Chloride in Snowmelt and Stormwater Runoff in Urban Areas				
Form of Runoff	**EMCs (mg/l)**	**# Events**	**Location**	**Source**
Snowmelt	116*	49	MN	Oberts, 1994
	2119	N/R**	Ontario	Sherman, 1998
	474	N/R	NY	Novotny *et al.*, 1999
	1612	N/R	WI	Masterson and Bannerman, 1994
	397	282	Ontario	Environment Canada, 2001
Stormwater runoff (non-winter)	42	61	TX	Brush *et al*, 1995
	45	N/R	Ontario	Sherman, 1998
	40.5	N/R	WI	Masterson and Bannerman, 1994

* = median
**N/R = Not Reported

Chloride concentrations found in roadside soils often exceed the tolerance thresholds of roadside wetland vegetation, and these elevated chloride levels have been documented up to 1,000 feet or more from the road (Kaushal *et al*., 2005; Wenger and Yaggi, 2001; EC, 2001; Richburg *et al*., 2001). Many plant species are sensitive to high chloride levels and may dieback or fail to germinate under these conditions (Biesboer and Jacobson, 1994). According to Environment Canada, more than 50% of woody plant species are sensitive to road salt and have disappeared from roadside wetlands and ditches (EC, 2001).

Wetland in warmer climates can also be impacted by chloride. Changes in wetland salinity due to evaporation following impoundment by water control structures have primarily been documented in arid regions or in coastal salt marshes impounded for mosquito control or wildlife management (ABAG, 1991; Wenner, 1986; Sinicrope *et al*., 1990). . Salinity fluctuations from impoundments cause changes in vegetation patterns and impacts to fish populations (USEPA, 1993; Gilmore *et al*., 1981; Sinicrope *et al*., 1990).

Chloride concentrations in wetlands in Northern climates have been detected as high as 2,700 mg/l.

Both acute and chronic toxic effects of chloride on aquatic systems have been well documented. Chronic concentrations of chloride as low as 210 mg/l have been found to be harmful to some forms of aquatic life. Chloride levels exceeding 1,000 mg/l can have lethal and sublethal effects on a wide range of aquatic plants and invertebrates (EC, 2001). Chloride interferes with a plant's ability to regulate water absorption, leading to dehydration (Friederici, 2004). Increases in chloride levels can lead to the spread of salt-tolerant plant species, many of which are undesirable (e.g., cattails and purple loosestrife). Chloride may also combine with heavy metals in wetland soils, rendering them more water soluble and more available for uptake by plant roots thereby possibly increasing the plant uptake of toxic metals (EC, 2001).

Numerous studies document chloride discharges to wetlands in northern latitudes, and the subsequent impacts to wetland plant and animal communities. Table 18 summaries these studies. Literature suggests that urban wetlands receiving excessive chloride will experience reduced biodiversity, a loss of sensitive species, and an increase in salt tolerant invasive species (EC, 2001).

Table 18. Research Review of Chloride Discharges to Wetlands

Key Finding(s)	Location	Reference
Average salinity in vernal pools within 650 feet of a road was seven times higher than average salinity in non-roadside pools. 60% of roadside pools had elevated salinity.	NY	Karraker and Gibbs, in review
Ephemeral wetlands adjacent to roads receiving salt had higher salt and chloride levels compared to wetlands not exposed to salt. Chloride concentrations approached lethal concentrations for hatching tadpoles, and were significantly higher in samples collected at points nearest the road at salted sites. Values remained elevated for 5 months, and may also have negative impacts on other wetland species.	MI	Murawski, 2005

Table 18. Research Review of Chloride Discharges to Wetlands		
Key Finding(s)	**Location**	**Reference**
Chloride concentrations in 43 road salt-impacted wetlands ranged from 18 to 2,700 mg/l with 75% less than 334 mg/l. Macroinvertebrate tolerance of chloride was found to be higher than concentrations in most of the wetlands.	MI	Benbow and Merrit, 2004
Decreases in community measures (richness, evenness, and overall cover) and individual species abundances in a fen were attributed to high salt concentration from nearby turnpike. Chloride concentrations were highest closest to the turnpike (210 to 275 mg/l) and gradually decreased with distance from the road. High chloride concentrations (> 54 mg/l) were present up to 980 feet from the turnpike.	MA	Richburg *et al*., 2001
In a greenhouse study, species diversity, richness, evenness, and total biomass all decreased with increasing snowmelt concentration. Common cattail and purple loosestrife were tolerant of snowmelt.	N/A	Isabelle *et al*.,1987
High concentrations of chloride in groundwater of two fens were linked to nearby road salt application and caused a loss of biodiversity. Diverse vegetation was replaced by the more salt-tolerant narrow-leaf cattail.	IL	Panno *et al*., 1999
Chloride concentrations in Atlantic white cedar wetlands in developed areas were elevated compared with control sites. Changes in vegetative community composition and structure were linked to this decline in water quality.	NJ	Ehrenfeld and Schneider, 1990
Contamination of a bog with road salt from a nearby salt storage area resulted in die-off of tamarack trees, red maples, sedges, pitcher plants, and sphagnum mosses. These species were replaced by more salt-tolerant species such as cattails. The highest concentrations of chloride in the wetland were 1,215 mg/l.	IN	Wilcox, 1986

Several wetland types are particularly vulnerable to high chloride levels in runoff. For example, wetlands lacking distinct outlets, such as vernal pools or prairie potholes, tend to accumulate chloride in the bottom where it cannot easily be flushed (Karraker, 2006). Atlantic white cedar wetlands exposed to road runoff containing chloride were reported to have decreased *Sphagnum* coverage and reduced cedar seedling numbers (Ehrenfeld and Schneider, 1990). Increases in chloride decreased *Sphagnum* coverage, thus resulting in decreased number of cedar seedlings. The decline of *Sphagnum* as a result of increasing runoff is shown in Figure 19. Road salt has also been associated with declines in native plant species, including *Sphagnum*, in an Indiana bog (Bubeck *et al.* 1971).

Similar declines in the fen plant community have also been attributed to high chloride levels in Massachusetts (Richburg *et al*., 2001). Species richness and plant cover were considerably lower in plots with high concentrations of sodium (112 – 267 mg/l) and chloride (54 -114 mg/l). Furthermore, dominant native fen species, such as hoary willow (*Salix candida*) and cranberry (*Vaccinium macrocarpon*) were significantly less abundant in plots with high levels of salt (Richburg *et al*., 2001). Dense stands of *Phragmites* were present throughout the fen; the authors concluded that as the dominant native vegetation declines in the salt impacted areas, *Phragmites* will continue to spread (Richburg *et al*., 2001). Table 19 lists certain wetland plant species that have been shown to be sensitive to road salt impacts.

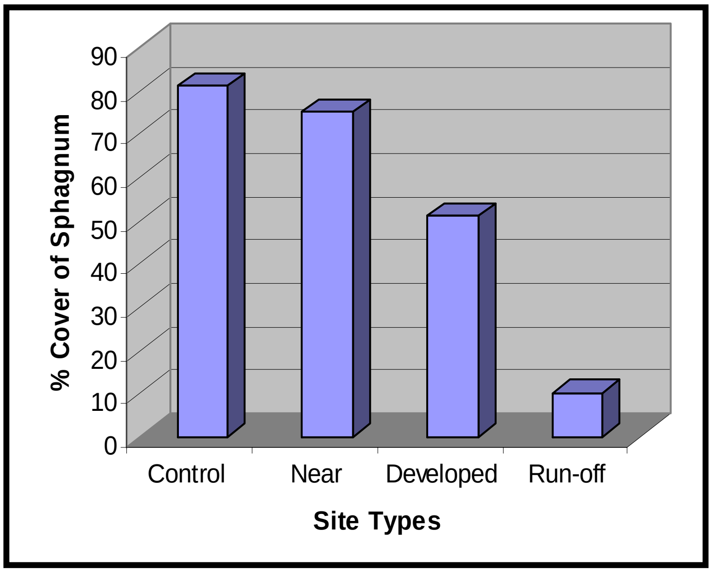

Figure 19. Mean percent cover of *Sphagnum* at each site type (compiled from Ehrenfeld and Schneider, 1991)

Table 19. Plant Species Sensitive to Runoff and Road Salt Impacts

Species affected	Location	Reference
• Atlantic white cedar (*Chamaecyparis thyoides*) • *Sphagnum* sp. • Tufted sedge *(Carex stricta)* • Round leaf sundew *(Drosera rotundifolia*) • Golden club (*Orontium aquaticum)* • Bog aster (*Aster nemoralis*) • Swamp pink (*Helonias bullata)* • Bladderwort *(Utricularia spp.)* • Cotton grass (*Eriophorum virginicum)* • Hazel alder (*Alnus serrulata*)	NJ	Ehrenfeld and Schneider, 1991
• Sedges (*Carex spp.)* • Red maple (*Acer rubrum*) • Pitcher plant (*Sarracenia purpurea)* • *Sphagnum* sp.	IN	Bubeck *et al.*, 1971
• Hoary willow (*Salix candida*) • Cranberry (*Vaccinium macrocarpon*)	MA	Richburg *et al.*, 2001

Several invasive species flourish when exposed to high chloride levels, and this can profoundly alter the wetland plant community. For example, researchers have noted that narrow-leaved cattail (*Typha augustifolia)* and common reed-grass (*Phragmites australis*) are frequently present in roadside swales and wetlands with high soil chloride levels (EC, 2001). Isabelle *et al.* (1987) determined that only two wetland plant species could germinate in pure roadside snowmelt conditions -- common cattail (*Typha latifolia*) and purple loosestrife (*Lythrum salicaria*). The tolerance of invasive species for both higher chloride levels and higher water level fluctuations allows them to rapidly establish and spread in wetlands that receive snowmelt and stormwater runoff (Isabella *et al*., 1987).

Chloride can also degrade the quality of the wetland invertebrate community. Gernes and Helgen (2002) sampled Minnesota wetlands exposed to stormwater runoff and found that the number of invertebrate taxa and intolerant taxa decreased as chloride concentrations increased in urban wetlands (Figure 20). Toxicity studies indicate that chloride becomes acutely toxic to wetland invertebrates at concentrations in the 2,500 to 4,500 mg/l range; this range was seldom achieved in the majority of Michigan wetlands sampled by Benbow and Merritt (2004). Toxicity data from Canada suggest an LC_{50} [4] for the crustacean *Ceriodaphnia dubia* is 1,400 mg/l, which may be exceeded in some wetlands adjacent to roadways and at snow disposal sites (EC, 2001). Sanzo and Hecnar (2006) reported that road salt had toxic effects on wood frog tadpoles in Ontario wetlands typified by lower survivorship and increased physical abnormalities.

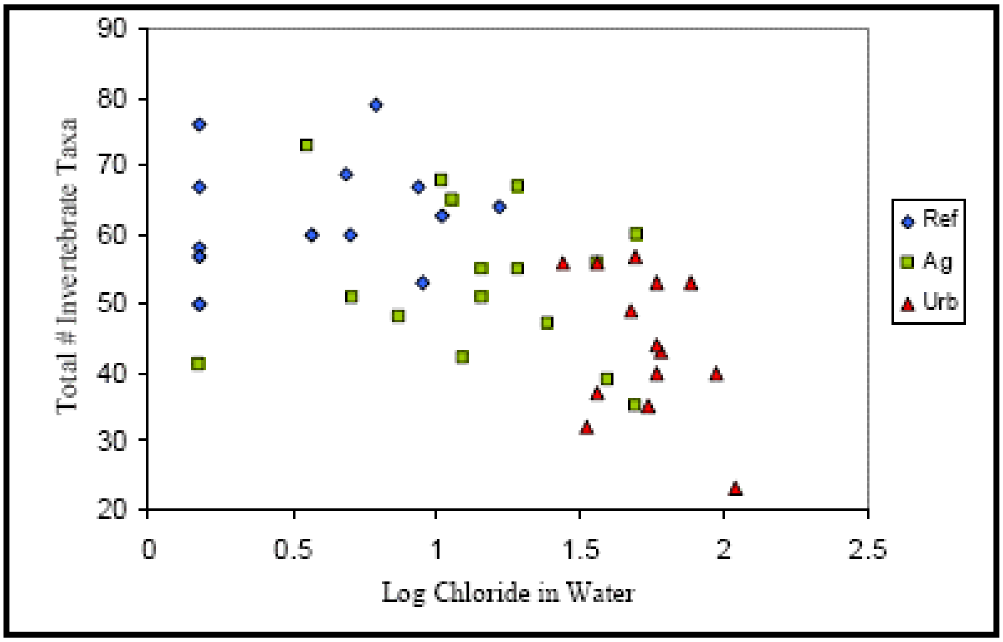

Figure 20. Effects of chloride on invertebrate abundance in Minnesota wetlands (from USEPA, 2002a)

Cumulative Impacts of Land Development on Wetland Communities

The preceding sections have reviewed how individual hydrologic and water quality stressors impact urban wetland quality. Most urban wetlands that are exposed to stormwater, however, are subject to a combination of different stressors. This section will review research on the cumulative impacts of land use change in the contributing drainage area on overall wetland quality. Cumulative impacts result in changes to habitat structure and ecosystem properties, which can have a domino effect on many plant and animal species in the wetland.

[4] LC_{50} denotes the "lethal concentration" at which 50% of the organisms die over a certain time period, which is four days in this study.

Cumulative Impact on the Wetland Plant Community

As might be expected, there are fewer studies that have examined the topic of cumulative impacts. Table 20 reviews the range of studies that have explored the cumulative impact of urbanization on the wetland plant community. The overall pattern is that changes in hydroperiod and pollutants influence the biological character of wetlands. The basic response is a decline in wetland plant diversity and a shift in plant community composition towards invasive and tolerant wetland plants.

Table 20. Recent Research on the Impacts of Urbanization on Wetland Plant Communities

Key findings	Location	Reference
Shrub-carr vegetation diversity and richness were highly correlated to land use at the 1,640 foot scale and diversity at the 3,280 foot scale.	MN	Mensing *et al.*, 1998
As watershed development increased, indigenous plant species declined and community structure was altered due to water quality changes and invasive species.	NJ	Ehrenfeld and Schneider, 1991
Watershed conditions were examined for Atlantic White cedar swamps across differing urban disturbance regimes. Sites with high disturbance had lower understory richness, high canopy cover of red maple, and low *Sphagnum* cover.	NJ	Laidig and Zampella, 1999
Land use changes between 1926 and 1988 from extensive agriculture to urban land resulted in distinct changes to the dominant vegetation in New York wetlands, specifically in emergent, forested, and scrub/shrub communities.	NY	Thibault and Zipperer, 1994
Urbanization caused changes in wetland hydroperiod, affecting plant communities (increase in exotic grasses) and species richness (50% decline over a few decades) within a wetland.	Ontario	Chow-Fraser *et al.*, 1998
Plant species richness decreased relative to increases in paved road density at all distances studied with the most impact on species richness found within 0.6 miles of the road.	Ontario	Findlay and Houlahan, 1997
Wetlands surrounded by urban land had more introduced species than wetlands surrounded by undeveloped land. Community composition was strongly related to percent cover of water for both wetland types.	OR	Magee *et al.*, 1999
The relative frequency of thin-stemmed emergent plants in Puget sound wetlands decreased over time due to increasing WLF and mean depth. WLF is linked strongly to IC in CDA.	WA	Chin, 1996
In both scrub/shrub and emergent wetlands, plant richness was significantly negatively correlated with percentage impervious area within the watershed and mean WLF.	WA	Reinelt *et al.,* 1998
Urbanization caused emergent meadows to become dominated by cattail or reed canary grass.	WA	Cooke and Azous, 1993
Large scale vegetation changes (increased in dominance by invasives and loss of native species) in emergent zones of palustrine wetlands due to urban runoff from land development changes.	WI	Owen, 1999

Cumulative Impacts on Aquatic Invertebrates

Aquatic invertebrates are some of the most commonly used indicators to measure the health of aquatic ecosystems (Karr and Chu, 1999; Rosenberg and Resh, 1993). Aquatic invertebrates are good biological indicators because they are easy to sample and they respond to many kinds of stressors over extended periods of time. Several researchers have designed invertebrate sampling protocols and metrics specifically geared to measure wetland quality (Azous and Horner, 1997; U.S. EPA, 2002b). Wetland invertebrates are an important element of the food web and typically spend most or all of their life cycle within wetlands. This direct and regular exposure to wetland conditions and stressors makes them an excellent indicator of wetland quality (USEPA, 2002b).

The relatively few studies published to date show that land development is linked to declining wetland invertebrate quality (Table 21). For example, Hicks and Larson (1997) found that the wetland invertebrate community was impaired beyond 20% impervious cover in the CDA, while wetlands with low CDA impervious cover (less than 5%) were found to be similar to natural reference wetlands. In studies of Minnesota wetlands, wetland invertebrate community measures, including total abundance and sensitive taxa, were inversely correlated to urbanization (Gernes and Helgen, 2002).

Table 21. Recent Research on the Relationship Between Urbanization and Aquatic Macroinvertebrate Communities

Key Finding(s)	Location	Reference
Macroinvertebrate species richness was greater in well-vegetated wetlands and abundance was greater in highly nutrient-enriched wetlands.	Australia	Balla and Davis, 1995
Changes in wetland hydrology and plant cover over a 50-year period have altered the benthic community in an Ontario wetland. Pollution-tolerant chironomids and oligochaetes and other worms have increased and pollution sensitive Trichopterans and Plecopterans have decreased.	Ontario	Chow-Fraser *et al*., 1998
An index of biological integrity was established by studying invertebrates in 44 natural, depressional, forested wetlands with varying levels of urbanization. Total abundance and species richness of sensitive taxa decreased as urbanization increased.	MN	Gernes and Helgen, 2002
Invertebrate community indices declined with increasing imperviousness (greater than 3%). Wetlands with more than 20% imperviousness had moderately to severely impaired habitat.	CT	Hicks and Larson, 1997

Cumulative Impacts on Amphibians and Reptiles

Amphibians are frequently cited as excellent indicators of wetland health given that they spend much of their life cycle in wetlands and select specific habitats on the basis of hydroperiod and other wetland conditions (Wake, 1991). Thus, amphibian populations are unusually attuned to wetland conditions and are extremely sensitive to alterations in wetland quality (US. EPA, 2002b). Less frequently, research has been conducted using reptiles as an indicator of wetland quality. Some reptiles such as turtles do spend critical parts of their life cycle in wetlands and rely on them for food and shelter. However, reptiles are less reliant on wetlands than amphibians.

The numerous studies that have linked urbanization to declines in amphibian abundance and richness in wetlands are profiled in Table 22. Early work by Richter and Azous (1995) found that wetlands with 40% urban land in their CDA had significantly lower amphibian richness than wetlands with less development in their CDA. Further work by Reinelt *et al*. (1998) in the same ecoregion found that the most urbanized wetlands had the lowest amphibian richness and the lowest proportion of native amphibian species.

Other researchers have reported correlations between forest cover (both in wetland buffers and upland areas) and amphibian and reptile populations in wetlands (Burke and Gibbons, 1995; Hecnar and M'Closkey, 1998; Knutson *et al.*, 1999; Semlitsch, 1998; Rubbo and Kiesecker, 2005). These studies suggest the need to link terrestrial forest habitats adjacent to wetlands to sustain amphibian and reptile species. For example, Burke and Gibbons (1995) demonstrated that three species of freshwater turtle utilized a 900-foot radius of upland habitat adjacent to a wetland for nesting and hibernation. The core terrestrial habitat of adult salamanders was estimated by Semlitsch (1998) to be approximately 500 feet from the boundary of a study wetland, while Semlitsch and Bodie (2003) suggest a range of 384 to 1207 feet, depending on the species (Table 23). These studies illustrate the need to protect wetlands and the upland buffers that surround them to sustain amphibian and reptile populations, especially in urbanized areas.

Table 22. Research on the Effects of Urbanization on Amphibians and Reptiles

Key Finding(s)	Location	Reference
Both richness and abundance were lower in a residential development than in an undeveloped park. Sensitive species decreased and tolerant species (e.g., bullfrog) increased. This trend is possibly related to the increase in permanent water.	FL	Delis *et al.*, 1996
Amphibian abundance and richness were negatively associated with the presence of urban land use. Positive associations were found between frogs/toads and upland and wetland forest.	IA, WI	Knutson *et al.*, 1999
Changes in hydrology due to development will impact bog turtle habitat.	MD	Brennan *et al.,* 2001
Amphibian abundance was significantly related to land use at both 1,640 feet and 3,280 feet.	MN	Mensing *et al.* 1998
Traffic density on roads within a radius of approximately one mile wetland ponds had a significant negative effect on leopard frog abundance, but not green frogs.	Ontario	Carr and Fahrig, 2001
Amphibian species richness and abundance were negatively correlated with road density near wetlands.	Ontario	Houlahan and Findlay, 2004
Urbanized wetlands had lower amphibian species richness – mainly loss of wood frogs and salamanders, which are negatively associated with hydroperiod and positively associated with amount of forest habitat.	PA	Rubbo and Kiesecker, 2005
Wetlands with urban area coverage >40% had lowest amphibian species richness; < 5% had high richness. Mean WLF < 7.9 inches had highest species richness.	WA	Richter and Azous, 1995
As impervious cover and WLF increase, amphibian richness and abundance declined.	WA	Chin, 1996

Table 22. Research on the Effects of Urbanization on Amphibians and Reptiles		
Key Finding(s)	**Location**	**Reference**
Decline in amphibian species richness as WLF increased above 8.7 inches in emergent and scrub/shrub wetland zones.	WA	Horner, *et al.* 1997a
Amphibian richness in wetlands was related to the degree of urbanization within the watershed. The most urbanized wetlands had the lowest species richness.	WA	Reinelt *et al*., 1998
In 30 wetlands, species richness of amphibians and reptiles was significantly related to the density of paved roads within 1.2 miles of the wetland edge.	Ontario	Findlay and Houlahan, 1997

Table 23. Core Terrestrial Habitat for Wetland Amphibians and Reptiles (from Semlitsch and Bodie, 2003)		
	Mean minimum in feet	**Mean maximum in feet**
Frogs	672	1,207
Salamanders	384	715
Amphibians	522	951
Snakes	551	997
Turtles	403	941
Reptiles	417	948
Herpetofauna	466	948

Other researchers note that the location of impervious cover in the CDA is important – roads located near wetlands are often a major source of amphibian and reptile mortality. Roads adjacent to wetlands can impact amphibian and reptile populations through direct mortality, reduced habitat access, and population fragmentation and isolation (Jackson, 2000). Amphibians and reptiles are vulnerable to road effects because their life histories involve migrating between wetlands and upland habitat (Ashley and Robinson, 1996; Trombulak and Frissell, 2000). Studies have shown that amphibian species richness declines with increasing road density near wetlands (Houlahan and Findlay, 2004; Findlay and Houlahan, 1997). Similarly, population abundance of the leopard frog (*Rana pipiens*) was negatively affected by traffic density within approximately a one-mile radius of the wetland or pond (Carr and Fahrig, 2001). Even roads with low traffic density produced high mortality for the American toad (*Bufo americanus*) and *Ranid* frogs (green, wood and leopard frogs) (Mazerolle, 2004). Aresco (2005) found that increases in traffic adjacent to wetlands has increased mortality of many turtles species.

Cumulative Impacts on Birds

Most bird species rely on wetland habitats during some portion of their life cycle. Consequently, birds may be directly and indirectly impacted by the degradation of wetland quality. Urbanization can degrade wetland habitats used for breeding, nesting or feeding, and change competitive interactions among and between species that modify populations (Richter and Azous, 1997a). For example, Mensing *et al*. (1998) and DeLuca *et al.* (2004) investigated the influence of adjacent land use on bird communities in riparian wetlands and estuarine wetlands.

They found that land development had a pronounced negative impact on wetland bird species richness at distances of 1,640 to 3,280 feet from the wetland. DeLuca *et al*. (2004) further concluded that land development covering as little of 14% of the area within 1,640 feet of a wetland was enough disturbance for certain bird species to abandon estuarine wetlands entirely. The proximity and density of roads can also have a significant effect on wetland bird communities.

Findlay and Houlahan (1997) concluded that roads and other forms of linear development reduced connectivity between wetland habitat patches, thereby reducing bird species richness. Table 24 presents a summary of research available on the relationship between wetland bird species and urbanization.

Richter and Azous (1997a) reported a correlation between watershed urbanization and declines in bird species richness as well as a higher number of non-native bird species. They also found that most of the bird species that were less tolerant of urbanization were found in forested areas within 1,640 to 3,280 feet of existing wetlands. Dowd (1992) investigated forested wetlands in New York that were surrounded by urban land and reported that they were dominated by non-forest, urban resident and human-attracted bird species. Likewise, DeLuca *et al.* (2004) found that wetlands located in heavily developed areas had bird communities with low species richness. Wetland specialist species, such as the least bittern (*Ixobrychus exilis)* were absent and were replaced by habitat generalists, such as red-winged blackbirds (*Agelaius phoeniceus)* and common grackles (*Quiscalus quiscula).*

Table 24. Recent Research on the Relationship Between Wetland Bird Species and Urbanization

Indicator	Key finding(s)	Location	Reference
Community Index	In Chesapeake Bay wetlands, development had a pronounced negative impact on bird integrity at both the 1,640 and 3,280 foot scales. A specific land development threshold of 14% was identified.	Chesapeake Bay	Deluca *et al*., 2004
Community composition	Bird species composition at a fragmented forested wetland consisted of more urban and non-forest species than a larger, unfragmented parcel.	NY	Dowd, 1992
Species Diversity	Diversity of birds was significantly negatively correlated with urban land use within 1,640 feet of the wetland.	MN	Mensing *et al*., 1998
Species richness	Bird species richness decreased relative to increases in paved road density at all distances studied with the most impact on species richness found in the first 1,640 feet out from the road.	Ontario	Findlay and Houlahan, 1997
Species richness	In Puget Sound palustrine wetlands, bird species richness decreased in developing watersheds. Within 0.6 miles of the wetland, diversity was influenced by urbanization.	WA	Richter and Azous, 1997a

Cumulative Impacts on Mammals

Few wetland mammals depend entirely on wetlands in North America (Gibbs, 1995). The majority of mammal species inhabit upland ecosystems as opposed to wetland areas (Adamus and Brandt, 1990). The most prominent wetland-dependent mammals include the muskrat (*Ondatra zibethicus*) and beaver (*Castor canadensis*). Other obligate mammals commonly found in wetlands include carnivorous shrews, lagomorphs, the swamp and marsh rabbits, mustelid, and river otter (Gibbs, 1995). Mammals that utilize wetlands extensively for feeding and cover, but also rely on upland habitat include raccoons, black bears, white-tailed deer, and moose (Gibbs, 1995; and May, 2001). Many wetland mammals are either herbivores or omnivores; consuming wetland plants directly or having a mixed animal-plant diet (Adamus and Brandt, 1990).

The distribution and abundance of small mammals can be indicative of the environmental health of wetlands. However, it is generally difficult to determine “normal” levels for parameters, such as mammal density, species richness, or biomass because quantitative data on the structure of the entire mammalian community of wetlands has not been uniformly collected in any region of the country. Information on the impacts to wetland mammals due to urbanization is limited mostly to studies of hydrologic effects and vegetation removal.

Several hypotheses and preliminary investigations exist regarding wetland mammal stressors due to urbanization. In terms of organic loading and acidification, the community composition is believed to shift from fish-eating species to vegetarian or invertebrate-eating species and opportunists. Another hypothesis is that as the distance between wetlands containing wetland-dependent mammals becomes greater and hydrologic connections and vegetated corridors become severed, the more sensitive mammals could be affected (Adamus and Brandt, 1990).

Several studies have found that species richness of small mammals in wetlands is positively correlated with the complexity of vegetation structure (Arner et al., 1976; Landin, 1985; Maki et al., 1980; Nordquist and Birney, 1980; Stockwell, 1985; Searls, 1974; and Simons, 1985). Small mammal communities change due to vegetation removal and den site destruction (Krapu et al., 1970; Malecki and Sullivan, 1987; and Possardt and Dodge, 1978). However, overstory removal increases the density of herbaceous ground cover, thereby increasing the abundance of small mammals (Adamus and Brandt, 1990).

Changes in wetland water levels and soil moisture also alter the community structure of mammals. During hibernation, the effects of dehydration can be severe due to exposure (Bellrose and Low, 1943). Mammals that inhabit subsurface areas are particularly sensitive to moisture level changes. However, local changes are not typically reflected by indicator species of mammals because they have the ability to move between impacted areas (Adamus and Brandt, 1990).

A study by Richter and Azous (1997b) focused on the distribution and abundance of small mammals across Puget Sound wetlands and compared wetland conditions to habitat characteristics important for maintaining diversity and unique species. They found that the percentage of forested land immediately adjacent to the wetland was positively correlated with mammal community diversity. The combined factors of wetland size, adjacent land use and the relative quantity of large woody debris within the wetland buffer were found to be associated with small mammal richness. Figure 21 shows this relationship. This suggests that a limited amount of development can occur if enough forest land remains available for cover, food, shelter, and microclimatic relief.

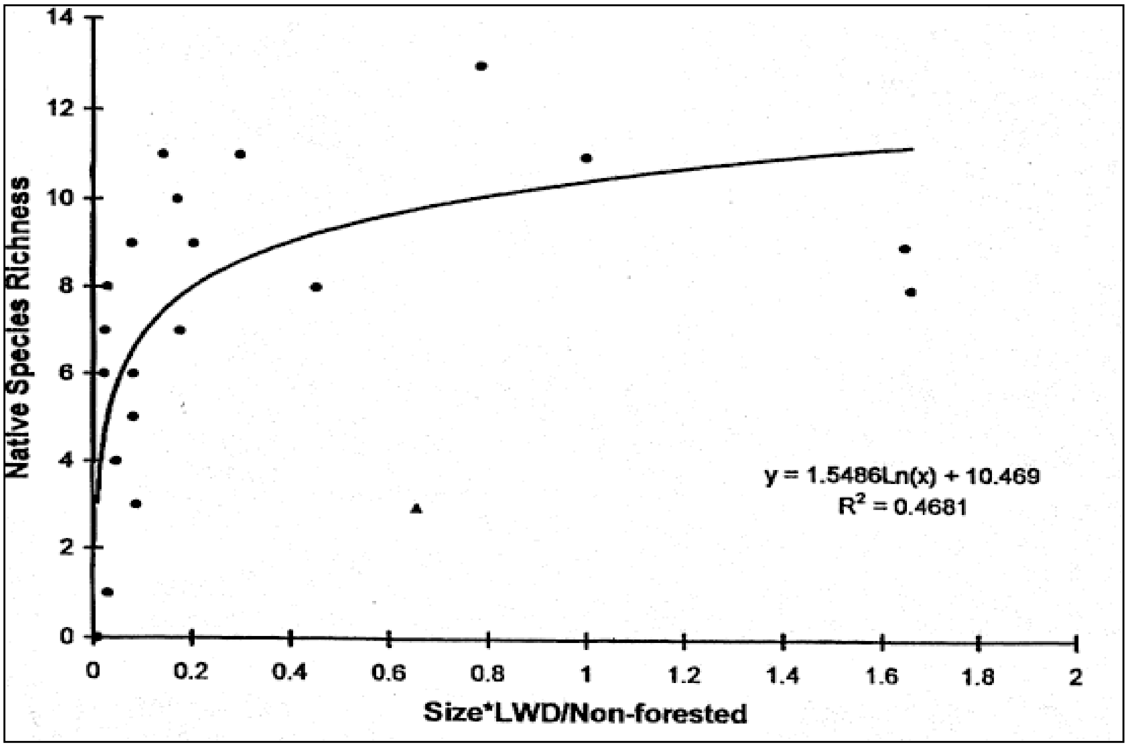

Figure 21. Relationship between small mammal species richness and habitat variables, including wetland size, land use cover, and large woody debris (Source: Richter and Azous, 1997)

Can CDA Impervious Cover Predict Cumulative Impacts?

While it is clear that even small amounts of land development can impair wetland quality, it is not yet possible to define a specific threshold of development that causes impairment. The most work to date is a collection of a dozen Puget Sound research studies that measured total impervious cover (IC) in the CDA to each wetland studied (e.g., Taylor et al., 1995; Chin, 1996; Horner et al. 1997a). One study from Connecticut also relates impervious cover thresholds to wetland quality (Hicks and Larson, 1997). These studies indicate a threshold of 3% to 5% IC where changes in wetland quality are detected and a second threshold at about 20% IC where wetland quality is sharply degraded. It is unclear, however, whether these findings from western Washington can be transferred or extended to other regions of the country.

Researchers in other parts of the country have not adopted the CDA impervious cover metric to define land development. Instead, they have used a diversity of alternative metrics such as percent urban land use, percent residential land use, road density, percent forest cover, and

percent developed land to characterize land use. In a few cases, researchers compute the development metric based on CDA characteristics, but it is more frequently computed based on adjacent land or distance from the wetland.
As a result, comparison among the research studies is difficult, and it is not currently possible to organize them into an overarching framework such as the Impervious Cover Model. The ICM is based on more than 150 research studies on the relationship between watershed impervious cover and various indicators of stream quality. As such, it has become a powerful tool for planners and engineers to predict future stream quality.

While it may not be possible to construct a wetland-based ICM at this time, it is clear that even low levels of development in the CDA can impact wetlands. Therefore, from a local wetland management standpoint, it makes sense to define and identify the wetlands most sensitive to land development in their CDA, and then regulate development activities within their CDAs to minimize indirect wetland impacts.

Designation of Sensitive Wetlands as an Alternative Management Approach

The research profiled in this article suggests that certain wetland types are sensitive to even low levels of land development or stormwater runoff. Other wetland types are less sensitive to adjacent land development, either due to their landscape position, plant communities, or the fact that they have already been degraded by urbanization or other disturbance. Table 25 presents a preliminary list of wetland types shown to be sensitive to urban stressors in the research profiled here. Communities should identify sensitive wetlands in their area as part of the local watershed planning process. Some states, such as Minnesota and New Hampshire, have designated wetland community types they consider sensitive to land disturbance (MNSWAG, 1997; Mitchell, 1996). Wetlands designated as sensitive should be afforded extra protection when development occurs in their CDA (see Article 3). Methods for defining and ground-truthing sensitive wetlands are presented in Article 2.

It is important to note that wetland sensitivity varies regionally, and a community should always develop its own locally-adapted list of sensitive and non-sensitive wetlands. As communities develop their lists, they should consult local wetland scientists to identify the wetland plant communities that have the greatest diversity, functional quality, and the least coverage of invasive plants. In general, high quality wetlands, or wetlands that contain rare, threatened, and endangered plant species should be automatically included on the list.

Table 25. Examples of Sensitive Wetlands	
Wetlands found to be sensitive in this literature review	
• Atlantic white cedar wetlands • Shrub-carr wetlands • Fens • Bogs • Vernal pools • Prairie potholes • Sedge meadows • Emergent meadows • Shallow marshes	• Ephemeral wetlands • Freshwater tidal wetlands • Shrub/scrub wetlands • Emergent wetlands • Forested wetlands • Palustrine wetlands • Headwater riparian wetlands • Depressional wetlands • Impoundments • Slope wetlands

Summary of Key Findings and Urban Wetland Research Gaps

This literature synthesis clearly documents that land development causes both direct and indirect impacts to wetlands that impair their function and quality. More systematic research is needed on the indirect impacts to wetlands, since it is extremely difficult to compare across wetland types, regions, plant communities, and landscape positions. In addition, it is not yet possible to directly link individual stressors generated in the CDA to predict impacts and biological responses within individual wetlands due to the interactions among many different stressors. With this in mind, several recommendations are provided to improve the future of urban wetland research.

- The current research on indirect impacts to wetlands has been produced by a great number of different academic disciplines that rarely interact with each other. Urban wetland research has been published by hydrologists, herpetologists, landscape ecologists, botanists, wildlife managers, conservation biologist, toxicologists, stormwater engineers and wetland scientists. It is recommended that a national meeting be convened or a network be launched to improve communication among the diverse research community currently working on the topic of indirect impacts to wetlands.

- Researchers have used many different metrics to describe the impact of upland development on wetlands (% urban land use, impervious cover, land cover, adjacent land use, forest cover, and percent developed). The lack of a uniform metric or index of land development as well as differences in how the CDA is defined and delineated has hindered comparison of studies. It is strongly recommended that researchers adopt a common convention for defining the CDA to wetlands and agree to measure a series of different land development metrics within the CDA.

- In addition, researchers may want to explore whether a common method (or methods) can be used to assess direct and indirect impacts to wetlands from the CDA in the field. The rapid Wetland Impact survey described in Article 2 of this series, may be a useful tool to start. Researchers should engage in a dialogue to develop more standard methods for monitoring and modeling hydrologic changes to wetlands.

- More systematic sampling of a large population of watersheds would be helpful in defining how watershed functions and indicators change in relation to percent wetland cover. This watershed-level wetland information could be important to help managers understand the importance of protecting wetlands to maximize watershed services.

- Perhaps the most critical research gap is the lack of understanding about wetlands whose water balance is dominated by groundwater, and more specifically, how these wetlands are impacted by upland changes in groundwater recharge rates due to land development. Although it is understandably difficult to track groundwater movement, more directed groundwater research is needed on this important topic.

- More research is warranted to explore how hydrologic changes, pollutants and other stressors promote the spread of invasive wetland plants. Current research indicates a general link, but does not yet indicate what causal factors can be manipulated by local wetland and watershed managers to reduce the spread of invasive species.

- A few studies have shown impacts to riparian wetlands due to stream constrictions. These studies are not necessarily conclusive as to the permanent effects on these wetlands, if any. Further research into the long-term impacts resulting from culverts, stream crossings, and other causes of flow constrictions is needed.

The research profiled here has shown that indirect impacts on wetlands from land development can have devastating and long-lasting impacts on many different wetlands, especially sensitive ones. This underscores the need for local protection of wetlands, since land use control is in the hands of local governments. A framework for using watershed planning to incorporate local wetland protection is provided in Article 2. Further tools available to local governments for protecting wetlands are specified in Article 3.

Local governments that wish to enact stronger local protection for wetlands and their functions can find a model ordinance to protect wetlands that are typically considered sensitive to stormwater runoff in Article 4. Article 5 deals with the topic of restoration, while the last article in the series addresses protection of small, isolated wetlands and other vulnerable aquatic resources that may not be fully protected by federal laws, especially in light of recent Supreme Court decisions.

References

ABAG. See Association of Bay Area Governments.

Acharya, G. 2000. Approaches to valuing the hidden hydrological services of wetland ecosystems. Ecological Economics 35: 63-74.

Adamus, P., and K. Brandt. 1990. Mammal Communities in *Impacts on Quality of Inland Wetlands of the United States: A Survey of Indicators, Techniques, and Applications of Community Level Biomonitoring Data.* Excerpts from Report #EPA/600/3-90/073. U.S. Environmental Protection Agency. Available Online at: http://www.epa.gov/owow/wetlands/wqual/mamm.html

Aresco, M.J. 2005. Mitigation Measures to Reduce Highway Mortality of Turtles and other Herpetofauna at a North Florida Lake. *Journal of Wildlife Management* 69(2): 549-560.

Arheimer B and H. Wittgren. 1994. Modelling the effects of wetlands on regional nitrogen transport. Ambio 23:378-386

Arner, D.H., Robinette, H.R. Frasier, J.E., and M.H. Grey. 1976. Effects of Channelization of the Luxapalila River on Fish, Aquatic Invertebrates, Water Quality and Furbearers. U.S. Fish and Wildlife Service. Washington, D.C., FWS/OBS-76-08.

Ashley, EP; Robinson, JT. 1996. Road Mortality of Amphibians, Reptiles and other Wildlife on the Long Point Causeway, Lake Erie, Ontario. *Canadian Field-Naturalist.* 110(3): 403-412.

Association of Bay Area Governments (ABAG). 1991. *Status and Trends Report on Wetlands and Related Habitats in the San Francisco Bay Estuary.* Third draft. Prepared for U.S. Environmental Protection Agency. Oakland, CA.

Azous, A.L., Reinelt, L.E., and J. Burkey. 1997. Managing Wetland Hydroperiod: Issues and Concerns. In Azous and Horner (eds) *Wetlands and Urbanization: Implication for the Future*. Washington State Department of Ecology. Olympia, WA.

Azous, A.L. and R.M. Horner. 2001. Wetlands and Urbanization: Implications for the Future. Lewis Publishers, Boca Raton, FL, USA.

Baird, C., T. Dybala, M. Jennings and D. Ockerman. 1996. *Characterization of Nonpoint Sources and Loadings to Corpus Christi National Estuary Program Study Area*. Corpus Christi Bay National Estuary Program. City of Corpus Christi, TX.

Bannerman, R., D. Owens, R Dodds, and N. Hornewer. 1993. Sources of Pollutants in Wisconsin Stormwater. *Water Science and Technology*. 28(3-5): 241-259

Barclay, E., D. Batker, and A. Kaler. 2004. *Untold Value: Natures Services in Washington State*. Prepared by the Asia Pacific Environmental Exchange. Seattle, WA. http://www.eartheconomics.org/documents/Natures_Services_WA_State.pdf

Baudo, R., Giesy, J., and Muntau, H. 1990. *Sediments: Chemistry and Toxicity of In-Place Pollutants*. Lewis Publishers, Inc. Chelsea, MI.

Barr, R. 1997. *Maryland NPDES Phase I Monitoring Data*. Maryland Department of the Environment. Baltimore, MD.

Bazemore, C. Hupp and T. Diehl. 1999. Wetland Sedimentation and Vegetation Patterns near Selected Highway Crossings in West Tennessee. USGS Water-Resources Investigations Report 91-4106.

Bellrose, F.C. Jr. and J.B. Low. 1943. The Influence of Flood and Low Water Levels on the Survival of Muskrats. *Journal of Mammals.* 24: 173-188.

Benbow, M. E., and R. W. Merrit. 2004. Road-Salt Toxicity of Select Michigan Wetland Macroinvertebrates under Different Testing Conditions. *Wetlands* 24(1):68-76.

Biesboer, D. D., and R. Jacobson. 1994. *Screening and Selection of Salt Tolerance in Native Warm Season Grasses*. Minnesota Department of Transportation. Report 94-11. 33 pp.

Bishop, C.A., Struger, J., Dunn, L., Forder, D.R. and Kok, S. 1999. *Stormwater Detention Ponds of Southern Ontario: Are they a Risk to Wildlife?* Minister of Public Works and Government Services Canada. Cat No.: En 40-222/9-1999E http://www.on.ec.gc.ca/wildlife/factsheets/fs_stormwater-e.html

Boyer, T and S. Polaski. 2004. *Valuing Urban Wetlands: A Review of Non-Market Valuation Studies*. Department of Applied Economics, University of Minnesota. St. Paul, MN. http://www.apec.umn.edu/faculty/spolasky/ValuingUrbanWetlands.pdf

Brennan, K. E., O'Leary, D. J., and S. P. Buckley. 2001. Hydrologic Analysis of the Wetland Habitat of the Federally Threatened Bog Turtle. *International Conference on Ecology and Transportation (ICOET) Proceedings 2001*. Available online at: http://repositories.cdlib.org/jmie/roadeco/Brennan2001a/.

Brinson, M. M. 1993a. *A Hydrogeomorphic Classification for Wetlands*. Technical Report WRP-DE-4. Waterways Experiment Station. Vicksburg, MS.

Brinson, M. M. 1993b. Changes in the Functioning of Wetlands along Environmental Gradients. *Wetlands* 13(2): 65-74.

Breaux, A.M., S. Farber and J. Day. 1995. "Using Natural Coastal Wetlands Systems or Wastewater Treatment: An Economic Benefit Analysis." *Journal of Environmental Management* 44:285-291.

Brock, T., S. J. H. Crum, R. van Wijngaarden, B. J. Budde, J. Tijink, A. Zuppelli, and P. Leeuwangh. 1992. Fate and effects of the insecticide Dursban 4E in indoor Elodea-dominated and macrophyte-free freshwater model ecosystems: I. Fate and primary effects of the active ingredient chlorpyrifos. Archives of Environmental Contamination and Toxicology 23:69-84.

Brush, S., Jennings, M., Young, J., and H. McCreath. 1995. NPDES Monitoring: Dallas-Ft. Worth, Texas Area. *Stormwater NPDES Related Monitoring Needs*. Proceedings of an Engineering Foundation Conference. Edited by Harry Torno. New York, NY.

Bryan, G.W. and Langston, W.J. 1992. Bioavailability, Accumulation and Effects of Heavy Metals in Sediment with Special Reference to United Kingdon Estuaries: a Review. *Environmental Pollutions* 76: 89-131.

Bubeck, R., W. Diment, B. L. Deck, A. Baldwin, and S. Lipton. *Runoff of Deicing Salt: Effect on Irondequoit Bay, Rochester, New York*. *Science* 11 172(3988):1128 - 1132

Burke, V. and J. Gibbons. 1995. Terrestrial Buffer Zones and Wetland Conservation: a Case Study of Freshwater Turtles in a Carolina bay. *Conservation Biology* 9: 1365-1369.

Burton, G.A., Nelson, M.K., Ingersoll, C.G. 1992. Freshwater Benthic Toxicity Tests. In Burton, G.A. (ed) *Sediment Toxicity Assessment.* Lewis Publishers, Inc. Chelsea, MI. Available for download at http://www.cerc.usgs.gov/pubs/sedtox/sedbib.htm

Cappiella et al. 2005. *Urban Watershed Forestry Manual Part 1: Methods for Increasing Forest Cover in a Watershed.* Center for Watershed Protection. Ellicott City, MD.

Carr, L.W. and L. Fahrig. 2001. Effect of Road Traffic on Two Amphibian Species of Differing Vagility. *Conservation Biology* 15(4): 1071-1078.

Carter, V. 1997. *Technical Aspects of Wetlands: Wetland Hydrology, Water Quality, and Associated Functions*. National Water Summary on Wetland Resources. U.S. Geological Survey Water-Supply Paper 2425. Available online at: http://water.usgs.gov/nwsum/WSP2425/hydrology.html

Carter, V. and Novitzki, R. 1988. Some Comments on the Relation Between Groundwater and Wetlands. Ch. 7 in Hook, P.D. *et al*. 1988. *The Ecology and Management of Wetlands.* Vol. 1. Timber Press, Portland, OR.

Casanova, M.T. and Brock, M.A. (2000). How do depth, duration and frequency of flooding influence the establishment of wetland plant communities? *Plant Ecology* 147: 237-250.

Center for Watershed Protection (CWP). 2003. *Impacts of Impervious Cover on Aquatic Systems*. Center for Watershed Protection. Ellicott City, MD.

Center for Watershed Protection (CWP). 1998. *Rapid Watershed Planning Handbook*. Center for Watershed Protection. Ellicott City, MD.

Cherry, D.S., Currie, R.J., Soucek, D.J., Latimer, H.A., and Trent, G.C. 2001. An Integrative Assessment of a Watershed Impacted by Acid Mine Discharges. *Environmental Pollution* 111: 377-388.

Chin, N. T. 1996. Watershed Urbanization Effects on Palustrine Wetlands: A Study of Hydrologic, Vegetative, and Amphibian Community Response Over Eight Years. Master's thesis, University of Washington.

Comer, P., K. Goodin, A. Tomaino, G. Hammerson, G. Kittel, S. Menard, C. Nordman, M. Pyne, M. Reid, L. Sneddon, and K. Snow. 2005. Biodiversity Values of Geographically Isolated Wetlands in the United States. NatureServe, Arlington, VA.

Chow-Fraser, Lougheed, V., LeThiec, V., Crosbie, B., Simser, L. and Lord, J. (1998). Long-term response of the biotic community to fluctuating water levels and changes in water quality in Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario. *Wetlands Ecology and Management,* 6: 19-42

Cooke, S.S. and Azous, A. 1997. The Hydrologic Requirements of Common Pacific Northwest Wetland Plant Species. In Azous and Horner (eds) *Wetlands and Urbanization: Implication for the Future*. Washington State Department of Ecology, Olympia, WA.

Cooke, S.S. and Azous, A. 1993. *Effects of Urban Stormwater Runoff and Urbanization on Palustrine Wetland Vegetation.* Puget Sound Wetlands and Stormwater Management Research Program Center for Urban Water Resources Management. University of Washington, Seattle, WA.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R., Paruelo, J., Raskins, R., Sutton, P., and M. Belt. 1997. The Value of the World's Ecosystem Services and Natural Capital. *Nature* 387: 253-260.

Craft, C.B., and W.P. Casey. 2000. Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA. Wetlands 20:323–332.

Crumpton,W. G. and L. G. Goldsborough. 1998. Nitrogen transformation and fate in Prairie wetlands. *Great Plains Research* 8:57-72.

Crunkilton, R., J. Kleist, J. Ramcheck, W. DeVita and D. Villeneuve. 1996. "Assessment of the Response of Aquatic Organisms to Long-term In Situ Exposures of Urban Runoff." *Effects of Watershed Development and Management on Aquatic Ecosystems.* Roesner, L.A. Editor. Proceedings of the ASCE Conference. Snowbird, Utah.

Culbertson, J.B., Valiela, I., Peacock, E., Reddy, C., Carter, A. 2005. *Continued Effects on the Salt Marsh Fiddler Crab,* Uca pugnax, *Following Long-term Petroleum Hydrocarbon Exposure.*

Conference abstract from the Estuarine Research Federation 2005 Conference. http://erf.org/user-cgi/conference05_abstract.pl?conference=erf2005&id=245

CWP. See Center for Watershed Protection.

Dahl, T.E. 2006. "Status and Trends of Wetlands in the Conterminous United States 1998 to 2004." U. S. Department of the Interior; Fish and Wildlife Service, Washington D.C.

Dahl, T. E. 2000. *Status and Trends of Wetlands in the Conterminous United States 1986-1997*. U.S. Department of the Interior, Fish and Wildlife Service. Washington, D.C.

Dahl, T. E. 1990. *Wetlands Losses in the United States 1780s to 1980s*. US Fish and Wildlife Service. Washington, DC.

Dahl, T. E. and C. E. Johnson. 1991. *Status and Trends of Wetlands in the Conterminous United States, Mid-1970s to Mid-1980s*. U.S. Fish and Wildlife Service.

Dartiguenave, C. M., I. ECLille, and D. R. Maidment. 1997. Water quality master planning for Austin, Texas. University of Texas at Austin Center for Research in Water Resources Online Report 97-6.

Davis, M. L. 1997. Statement of M. L. Davis, Deputy Assistant Secretary of the Army (Civil Works) before the Transportation and Infrastructure Committee, Subcommittee on Water Resources and the Environment. United States House of Representatives. April 29, 1997.

Delis, P.R., Mushinsky, H. R., and E.D. McCoy. 1996. Decline of some West-Central Florida Anuran Populations in Response to Habitat Degradation." *Biodiversity and Conservation* 5(2): 1579-1595.

DeLuca, W.V., Studds, C.E., Rockwood, L.L., and Marra, P.P. 2004. Influence of Land Use on the Integrity of Marsh Bird Communities of Chesapeake Bay, USA. *Wetlands* 24(4): 837-847.

Demers, C. L., and R. W. Sage, Jr. 1990. Effects of Road De-icing Salt on Chloride Levels in four Adirondack Streams. *Water, Air and Soil Pollution* 49: 369-373.

Dowd, C. 1992. *Effect of Development on Bird Species Composition of Two Urban Forested Wetlands in State Island, NY.* J. Field Ornitol. 63(4):455-461.

DU. See Ducks Unlimited.

Dubinski, B.J., R.L. Simpson, and R.E. Good. 1986. The retention of heavy metals in sewage sludge applied to a freshwater tidal wetland. *Estuaries* 9:102–111.

Du Laing, G., N. Bogaert, F.M. Tack, M.G. Verloo, and F. Hendrick. 2002. Heavy Metal Contents (Cd, Cu, Zn) in Spiders (Pirata piraticus) Living in Intertidal Sediments of the River

Scheldt Estuary (Belgium) as Affected by Substrate Characteristics." *Science of the Total Environment* 289(1-3): 71-81.

Ducks Unlimited (DU). No Date. *Groundwater Recharge*. Available online at: http://www.ducks.org/Page1547.aspx accessed August 2, 2006

Dunbabin, J.S. and Bowmer, K.H. 1992. Potential Use of Constructed Wetlands for Treatment of Industrial Waste Waters Containing Metals. *Science of the Total Environment* 111(2-3): 151-168.

EC. See Environment Canada.

Ehrenfeld, J. G. 2000. Evaluating Wetlands within an Urban Context. *Urban Ecosystems* 4: 69-85.

Ehrenfeld, J.G. and J.P. Schneider. 1990. The Response of Atlantic White Cedar Wetlands to Varying Levels of Disturbance from Suburban Development in the New Jersey Pinelands. In D.F. Whigham et al. (eds). *Wetland Ecology and Management: Case Studies*, 63-78. Kluwer Academic Publishers.

Ehrenfeld, J.G. and J.P. Schneider. 1991. *"Chamaecyparis thyoides* Wetlands and Suburbanization: Effects on Hydrology, Water Quality and Plant Community Composition." *J. Applied Ecology*. 28:467- 490.

Ehrenfeld, J.G. and J.P. Schneider. 1990. "The Response of Atlantic White Cedar Wetlands to Varying Levels of Disturbance from Suburban Development in the New Jersey Pinelands." *Wetland Ecology and Management: Case Studies 63-77.* (Ed. by D.F. Whigham, R.E. Good and J. Kvet).

Ellison, A. M., and B. L. Bedford. 1995. Response of a wetland vascular plant community to disturbance: a simulation study. *Ecological Applications* 5: 109-123.

ELI. See Environmental Law Institute.

Envirocast. 2003. April Showers Bring Muddy Waters: The Sour and Power of Raindrops. Available at http://www.stormcenter.com/envirocast/2003-04-01/envirocast-article1.php.

Environment Canada (EC). 2001. *Assessment Report: Road Salt*. Available online at: http://www.ec.gc.ca/substances/ese/eng/psap/final/roadsalts.cfm. Accessed August 30, 2006.

Environment Canada (EC). 2001. *Priority Substances List Assessment Report: Road Salts*. Canadian Environmental Protection Act, 1999

Environmental Law Institute (ELI). 2005. *State Wetland Program Evaluation Phase I.* Environmental Law Institute. Washington, DC.

EOA, Inc. 2001. *Stormwater Environmental Indicators Pilot Demonstration Project*. Final Report. Water Environment Research Foundation. Santa Clara Urban Runoff Pollution Prevention Project. Santa Clara, CA.

Ernst, K.A., Brooks, J.R., 2003. Prolonged flooding decreased stem density, tree size and shifted composition towards clonal species in a central Florida hardwood swamp. Forest Ecology and Management 173, 261–279.

Euliss, N. H., Jr. and D. M. Mushet. 1996. Water-level fluctuation in wetlands as a function of landscape condition in the prairie pothole region. Wetlands 16:587-593.

Ewel, K.C. 1990. Multiple Demands on Wetlands. *BioScience*. 40: 660-666.

Faber, S. and R. Costanza. 1987. The Economic Valuation of Wetland Systems. *Journal of Environmental Management*. 24: 41-51.

Fernandes, L.,M. A. Warith, and F. La Forge. 1996. Modelling of contaminant transport within a marshland environment. *Waste Management* 16:649-661.

Field, R., and R. Pitt (1990). Urban storm-induced discharge impacts: US Environmental Protection Agency research program review. Water Science and Technology.

Findlay, C. S. and Houlahan, J. 1997. Anthropogenic Correlates of Species Biomass Richness in Southeastern Ontario Wetlands. *Conservation Biology*, 11(4): 1000-1009.

Forman, R.T.T. and R.D. Deblinger. 2000. The Ecological Road-Effect Zone of a Massachusetts (U.S.A.) Suburban Highway. *Conservation Biology*, 14(1): 36-46.

Frayer, W. E. Monahan, T. J., Bowden, D. C., and F. A. Graybill. 1983. *Status and Trends of Wetlands and Deepwater Habitats in the Conterminous United States, 1950's to 1970's*. Colorado State University, Fort Collins, CO.

Friederici, P. 2004. Salt on the Earth. How Snow Control has Created High-Sodium Landscapes that Favor a Few Invasive Species. *Chicago Wilderness Magazine*. Winter 2004.

Gabor, T. A. North, L Ross, H. Murkin, J. Anderson, and M. Raven.2004. *Natural Values. The Importance of Wetlands and Upland Conservation Practice in Watershed Management: Functions and Values for Water Quality and Quantity*. Available online at: http://www.ducks.ca/conserve/wetland_values/pdf/nvalue.pdf

Gaiser, E.E., L.J. Scinto, J.H. Richards, K. Jayachandran, D.L. Childers, J.D. Trexler, and R.D. Jones. 2004. Phosphorus in periphyton mats provides the best metric for detecting low-level P enrichment in an oligotrophic wetland. Water Res. 38:507–516.

Galbraith, J., Donovan, P., Pantaleoni, E., and A. Sandy. 2005. *Density of Road Crossings and Impacts on Wetlands in Rural and Suburban Study Areas in the Coastal Plain of Virginia.*

Abstract, Powerpoint, and speaker notes, Virginia Tech Department of Crop, Soil, and Environmental Sciences.

Garn, 2002. *Effects of Lawn Fertilizer on Nutrient Concentrations in Runoff from Lakeshore Lawns, Lauderdale Lakes, WI*. USGS Water Resources Investigations Report 02-4130.

Gernes, M.C. and J.C. Helgen. 2002. *Indexes of Biological Integrity (IBI) for Large Depressional Wetlands in Minnesota*. Minnesota Pollution Control Agency, St. Paul, MN.

Gibbs, R.J. 1973. Mechanisms of Trace Metal Transport on Rivers. *Science*. 180(71): 71-73.

Gibbs, J.P. 1995. Hydrologic Needs of Wetland Mammals. *Encyclopedia of Environmental Biology*. 2: 267-276.

Gilmore R.G., D.W. Cooke, and C.J. Donohoe. 1981. A Comparison of the Fish Populations and Habitat in Open and Closed Salt Marsh Impoundments in East-Central Florida. *Northeast Gulf Science* 5(2): 25-30.

Gleason, R.A., Euliss, N.H. Jr., Hubbard, D.E. and Duffy, W.G. (2003). Effects of Sediment Load on Emergence of Aquatic Invertebrates and Plants from Wetland Soil Egg and Seed Banks. *Wetlands*, 23(1): 26-34.

Gregory, R. and K. Wellman. 2001. “Bringing stakeholder values into environmental policy choices: A community-based estuary case study.” *Ecological Economics* 39: 37-52. http://yosemite.epa.gov/SAB/sabcvpess.nsf/0/26bf3631d4a4303985256e4c0060450d/$FILE/040126124040170968.pdf

Groffman, P.M., D.J. Bain, L.E. Band, K. T. Belt, G.S. Brush, J.M. Grove, R. V. Pouyat, I.C. Yesilonis and W. C. Zipperer. 2003. Down by the riverside: Urban riparian ecology. Frontiers in Ecology and Environment 6:315-321.

Gwin, S.E., Kentula, M.E. and P.W. Shaffer. 1999. Evaluating the Effects of Wetland Regulation through Hydrogeomorphic Classification and Landscape Profiles. *Wetlands* 19(3): 477-489.

Hayashi, M., van der Kamp, G., and R. Schmidt. 2003. Focused Infiltration of Snowmelt Water in Partially Frozen Soil under Small Depressions. *Journal of Hydrology* 270: 214-229. Available online at: http://www.geo.ucalgary.ca/~hayashi/glgy699_17/papers/Hayashi2003.pdf

Hecnar, S. and M’Closkey, R. 1998. Species Richness Patterns of Amphibians in Southwestern Ontario Ponds. *Journal of Biogeography* 25: 763-772.

Heimlich, R., K. Wiebe, R. Claassen, D. Gadsby, and R. House. 1998. *Wetlands and Agriculture: Private Interests and Public Benefits.* Resource Economics Division, Economic Research Service, U.S. Department of Agriculture. Agricultural Economic Report No. 765. Available online at: http://www.ers.usda.gov/publications/aer765/

Hey, D. L. and N. S. Philippi. 1995. Flood reduction through wetland restoration: The upper Mississippi River basin as a case study. *Restoration Ecology* 3:4-17.

Hicks, A.L. and J.S. Larson. 1997. Aquatic invertebrates as an index for estimating the impacts of urbanization on freshwater wetlands. The Environmental Institute, University of Amherst, MA. Report submitted to U.S. Environmental Protection Agency, Corvallis, OR.

Horner, R. R., Booth, D. B., Azous, A., and C. W. May. 1997a. Watershed Determinants of Ecosystem Functioning. In Roesner, L. A. (Ed). *Effects of Watershed Development and Management on Aquatic Ecosystem*, Proceedings of the ASCE Conference. Snowbird, UT, 1996.

Horner, R.R.; Cooke, S.S.; Reinelt, L.E.; Ludwa, K.A.; and Chin, N.T. 1997b. Water Quality and Soils. In Azous, A. and Horner, R. (eds). *Wetlands and Urbanization: Implications for the Future*. Washington State Department of Ecology, Olympia, WA.

Horner, R. , J. Guedry, and M. Kortenhoff. 1990. *Improving Cost-Effectiveness of Construction Site Erosion and Pollution Control*. Washington State Department of Transportation. WA-RD-200.1. Seattle, WA.

Houlahan, J.E. and C.S. Findlay. 2003. The Effects of Adjacent Land Use on Wetland Amphibian Species Richness and Community Composition. *Can J. Fish Aquat. Sci.* 60: 1078-1094.

Imus, W. 2003. *Wetland and River Restoration Costs: Yesterday and Today*. Presented at the Headwaters to Ocean Conference, Long Beach, CA. Available online at: http://www.coastalconference.org/h20_2004/pdf/C-Sessions/F3C/Imus_H2O%20Conference.Session%203C.pdf

Isabelle, PS; Fooks, LJ; Keddy, PA; Wilson, SD. 1987. Effects of Roadside Snowmelt on Wetland Vegetation: An Experimental Study. *Journal of Environmental Management*. 25(1): 57-60.

Jackson, S.D. 2000. Overview of Transportation Impacts on Wildlife Movement and Populations. In Messmer, T.A. and B.West, (eds). *Wildlife and Highways: Seeking Solutions to an Ecological and Socio-Economic Dilemma*, The Wildlife Society. p. 7-20.

John, D.A and Leventhal, J.S. 1995. Bioavailability of Metals. In du Bray, E. (Ed). *Preliminary Compilation of Descriptive Geoenvironmental Mineral Deposit Models* U.S. Geological Survey, Open-File Report 95-831. Denver, Colorado. Available online at: http://pubs.usgs.gov/of/1995/ofr-95-0831/NAVIGATE.pdf

Karr, J. R., and E.W. Chu. 1999. *Restoring Life in Running Waters: Better Biological Monitoring*. Island Press, Washington, D. C.

Karraker, N. 2006. Road De-Icers used Near Wetlands: are they Worth their Salt? *National Wetlands Newsletter.* 28(1): 15-19.

Karraker, N. E., and J. P. Gibbs. In Review. Impacts of Road De-Icing Salt on the Demography of Vernal Pool-Breeding Amphibians. *Ecological Applications*.

Kaushal, S., Groffman, P., Likens, G., Belt, K., Stoack, W., Kelly, V., Band, L., and G. Fisher. 2005. Increased Salinization of Fresh Water in the Northeastern United States. *Proceedings of the National Academy of Sciences*. 102(38): 13517-13520.

Kercher, S., Carpenter, Q.J., and J. Zedler. 2004. Interrelationships of Hydrologic Disturbance, Reed Canary Grass (*Phalaris arundinaces* L.), and Native Plants in Wisconsin Wet Meadows. *Natural Areas Journal*. 24: 316-325.

King, R.S., Nunnery, K.T., and Richardson, C.J. (2000). Macroinvertebrate Assemblage Response to Highway Crossings in Forested Wetlands: Implications for Biological Assessment. *Wetlands, Ecology and Management*. 8: 243-256.

Kiviat, K. 2004. The Hackensack Meadowlands: History, Ecology, and Restoration of a Degraded Urban Wetland. *Urban Habitats*. 2(1). Available online at: http://www.urbanhabitats.org/v02n01/introduction.html

Klein, R. 1979. Urbanization and Stream Quality Impairment. *Water Resources Bulletin* 15(4): 948-963.

Knutson, M. G., Sauer, J. R., Olsen, D. A., Mossman, M. J., Hemesath, L.M., and Lannoo, M.J. 1999. Effects of Landscape Composition and Wetland Fragmentation on Frog and Toad Abundance and Species Richness in Iowa and Wisconsin, USA. *Conservation Biology*. 13(6): 1437-1446.

Koning, C. 2004. Impacts of Small Amounts of Sandy Sediment on Wetland Soil and Vegetation: Results from Field and Greenhouse Studies. *Wetlands*. 24(2): 295-308.

Krapu, G.L., D.R. Parsons, and M.W. Weller. 1970. Waterfowl in Relation to Land Use and Water Levels on the Spring Run Area. Iowa St. *Journal of Science*. 44: 437-452.

Krieger, D. 2001. *Economic Value of Forest Ecosystem Services: A Review*. The Wilderness Society. Available online at: http://www.wilderness.org/Library/Documents/upload/Economic-Value-of-Forest-Ecosystem-Services-A-Review.pdf

Kusler, J. 2003. *A Guide for Local Governments: Wetlands and Watershed Management*. DRAFT. McKnight Foundation, USEPA, National Parks Service. 153 pp.

Laidig, K. J. and R. A. Zampella. 1999. Community attributes of Atlantic white cedar (Chamaecyparis thyoides) swamps in disturbed and undisturbed Pinelands watersheds. *Wetlands* 19:35-49.

Landin, M.C. 1985. Bird and Mammal Use of Selected Lower Mississippi River Borrow Pits. Ph.D. Diss., Mississippi State Univ., MS.

Legg, A, R. Bannerman and J. Panuska. 1996. *Variation in the Relation of Rainfall to Runoff from Residential Lawns in Madison, WI*. USGS Water Resources Investigation Report 96-4194.

Lepp, N.W. 1981. *Effects of Heavy Metal Pollution in Plants (Vol. 1)*. Applied Sciences publication, New Jersey.

Leschine, T., Wellman, K. and Green, T. 1997. The Economic Value of Wetlands. Ecology Publication No. 97-100. Washington State Department of Ecology. Bellevue, Washington. Available online at: www.ecy.wa.gov/pubs/97100.pdf

Levine, J.F., Bogan, A.E., Pollock, K.H., Devine, H.A., Gustafson, L.L., Eads, C.B., Russell, P.P., and Anderson, E.F. 2003. Distribution of Freshwater Mussel Populations in Relationship to Crossing Structures. The Center for Transportation and the Environment/N.C. Department of Transportation, Report No. HWY-2003-02.

Mahaney, W.M., Wardrop, D.H. and Brooks, R.P. 2004. Impacts of Sedimentation and Nutrient Enrichment on Wetland Plant Community Development. *Plant Ecology*. 175: 227-243.

Mahler, B.J., Van Metre, P.C., Bashara, T.J., Wilson, J.T., and Johns, D.A. 2005. Parking Lot Sealcoat: an Unrecognized Source of Urban Polycyclic Aromatic Hydrocarbons. *Environmental Science & Technology* 39(15): 5560-5566.

Maki, T.E., D.W. Hazel, and A.J. Weber. 1980. Effects of Stream Channelization on Bottomland and Swamp Forest Ecosystems. North Carolina State Univ., Raleigh, NC. 66 pp. (NTIS PB-269 021/2St).

Malecki, R.A. and J.D. Sullivan. 1987. Assessment of an Agricultural Drainage Improvement Program in New York State. *Journal of Soil and Water Conservation.* 42: 271-276.

Martin, D and R. Neely. "Benthic macroinvertebrate response to sedimentation in a *Typha augustofolia* L. wetland." Wetlands Ecology and Management. 9:441-554.

Masterson, J. and R. Bannerman. 1994. *Impact of Stormwater Runoff on Urban Streams in Milwaukee County, Wisconsin*. Wisconsin Department of Natural Resources. Madison, Wisconsin.

May, H. 2001. Wetland Mammals. *Fish and Wildlife Habitat Management Leaflet Number 21*. Natural Resources Conservation Service. Available Online at: http://www.sc.nrcs.usda.gov/intranet/Dick%20Yetter%20Information/Tech%20Notes%20for%20Web%20Site/Biology%20Tech%20Note_21_WetlandMammals.pdf

May, Christopher, R. Horner, J. Karr, B. Mar, and E. Welch. 2000. Effects of Urbanization on Small Streams in the Puget Sound Ecoregion. In Schueler and Holland (eds). *The Practice of Watershed Protection*. Center for Watershed Protection. Ellicott City, MD.

Mazerolle, M. J. 2004. Amphibian Road Mortality in Response to Nightly Variations in Traffic Intensity. *Herpetologica* 60(1): 45-53.

Menzie-Cura & Associates. 1995. *Measurements and Loadings of Polycyclic Aromatic Hydrocarbons (PAH) in Stormwater, Combined Sewer Overflows, Rivers, and Publicly Owned Treatment Works (Pots) Discharging to Massachusetts Bays.* Report to the Massachusetts Bay Program, August 1995, MBP-95-06.

Mensing, D.M., Galatowitsch, and J. R. Tester. 1998. Anthropogenic Effects on the Biodiversity of Riparian Wetlands of a Northern Temperate Landscape. *Journal of Environmental Management*, 53: 349-377.

Miller, R. C. and J. B. Zedler. 2003. Responses of Native and Invasive Wetland Plants to Hydroperiod and Water Depth*.* *Plant Ecology* 167: 57-69.

Mitchell, F. 1996. Vegetated buffers for wetlands and surface waters: Guidance for New Hampshire and municipalities. *Wetland Journal* 8(4): 4-8.

Mitsch, W. and J. Gosslink. 2000. The Values of Wetlands: Importance of Scale and Landscape Setting. *Ecological Economics*. 35(200): 25-33. Available online at: http://swamp.ag.ohio-state.edu/aEcological%20Economics.pdf

Mitsch, W. J., B. C. Reeder, and D. M. Klarer. 1989. The role of wetlands in the control of nutrients with a case study of western Lake Erie. In: *Ecological Engineering: An Introduction to Ecotechnology*. (W. J. Mitsch and S. E. Jørgensen, eds.) pp. 129-158.Wiley, New York.

Mitsch, W.J., and J. Gosselink. 1993. *Wetlands*. Van Nostrand Reinhold, New York. 722 pp.

MN SWAG. See State of Minnesota Storm-Water Advisory Group (MNSWAG).

Moring, J. and D. Rose. 1997. Occurrence and Concentrations of Polycyclic Aromatic Hydrocarbon in Semipermeable Membrane Devices and Clams in Three Urban Streams of the Dallas-Fort Worth Metropolitan Area, Texas. *Chemosphere* 34(3): 551-566.

Mostaghimi, T., M, Wynn, J. Frazee, P. McClelland, R. Shaffer and W. Aust. 1994. *Effects of Forest Harvesting Best Management Practices on Surface Water Quality in the Virginia Coastal Plain.* Transaction of the American Society of Agricultural Engineers. 43(4): 927-936

Murawski, J. L. 2005. In press. *The Impacts of Deicing Agents on the Water Chemistry of Wetlands Adjacent to Roads in Four Mid-Michigan Counties*.

National Research Council (NRC). 2001. Compensating for Wetland Losses Under the Clean Water Act. National Academy Press. Washington D.C.

National Resources Defense Council (NRDC). 2003. *Bush Administration Plans to Limit Scope of Clean Water Act.* Press Release. Available online at: https://www.nrdc.org/media/pressreleases/030203.asp

Nordquist, G.E. and E.C. Birney. 1980. The Importance of Peatland Habitats to Small Mammals in Minnesota. Minnesota Department of Natural Resources, St. Paul.

Novitzki, R. P. 1979. Hydrologic characteristics of Wisconsin's wetlands and their influence on floods, stream flow, and sediment. In: *Wetland Functions and Values: The State of our Understanding.* (P. E. Greeson, J. R. Clark, and J. E. Clark, eds.) pp. 377-380. American Water Resources Association, Minneapolis, Minnesota.

Novotny, V., Smith, D. W., Kuemmel, D. A., Mastriano, J., and A. Bartosova. 1999. *Urban and Highway Snowmelt: Minimizing the Impacts on Receiving Water.* Water Environment Research Foundation. Alexandria, VA.

NRC. See National Research Council.

NRDC. See National Research Defense Council.

Oberts, G. 1994. Influence of Snowmelt Dynamics on Stormwater Runoff Quality. *Watershed Protection Techniques* 1(2): 55-61.

O'Brien, A.L. 1988. Evaluating the Cumulative Effects on New England Wetlands. *Environ. Manag.* 12(5): 627-636.

Olewiler, N. 2004. The Value of Natural Capital in Settled Areas of Canada. Published by Ducks Unlimited Canada and the Nature Conservancy of Canada. 36pp. Available online at: http://www.ducks.ca/aboutduc/news/archives/pdf/ncapital.pdf

Owen, C. 1999. Hydrology and History: Land Use Changes and Ecological Responses in an Urban Wetland. *Wetlands Ecology and Management*. 6(4): 209-219.

Panno, S. V., Nuzzo, V. A., Cartwright, K., Hensel, B. R., and I. G. Krapac. 1999. Impact of Urban Development on the Chemical Composition of Groundwater in a Fen-Wetland Complex. *Wetlands* 19(1): 236-245.

Pitt, R. 1987. *Small Storm Urban Flow and Particulate Washoff Contribution to Outfall Discharge*. PhD dissertation. University of Wisconsin, Madison.

Pitt, R.., A. Maestre, R. Morquecho, T. Brown, T. Schueler, K. Cappiella, P. Sturm, and C. Swann. *National Stormwater Q quality Database*. Center for Watershed Protection. Ellicott City, MD.

Possardt, E.E. and W.E. Dodge. 1978. Stream Channelization Impacts on Songbirds and Small Mammals in Vermont. *Wildlife Society Bulletin* 6: 18-24.

Reinelt, L. E. and R. R. Horner. 1991. Urban Stormwater Impacts on the Hydrology and Water Quality of Palustrine Wetlands in the Puget Sound Region. *Puget Sound Research. '91 Proceedings*. Puget Sound Water Quality Authority. 1: 33-42.

Reinelt, L.E. and B.L. Taylor. 2001. Effects of watershed development on hydrology. Chapter 8, pages 221-236 in A.L. Azous and R.R. Horner (eds.), Wetlands and Urbanization: Implications for the Future. Lewis Publishers.

Reinelt, L., Horner, R., and Azous, A. 1998a. Impacts of Urbanization on Palustrine (Depressional Freshwater) Wetlands – Research and Management in the Puget Sound Region. *Urban Ecosystems*, 2(4): 219- 236.

Reinelt, L., Horner, R., and Azous, A. 1998b. Impacts of Urbanization on Palustrine (Depressional Freshwater) Wetlands – Research and Management in the Puget Sound Region. *Urban Ecosystems*, 2(4): 229. Figure 5a.

Resh, V.H. and Rosenberg, D.M. (ed). 1984. *The Ecology of Aquatic Insects.* Praeger Publishers, New York, NY.

Richard, D. E., and D. C. Connell. 2001.Wetland restoration remediates chlorinated solvents in groundwater and protects surface water. Phytoremediation, Wetlands and Sediments: The Sixth International In Situ and On-Site Bioremediation Symposium. A. Leeson, E. A. Foote, M. K. Bankds, and V. Magar, eds. Battelle Press, Columbus, Ohio.

Richardson, C.J. 1995. Wetlands Ecology. In W. Nierenberg (ed.) *Encyclopedia of Environmental Biology*. Academic Press. San Diego, CA.

Richardson, C.J., Flanagan, N.A., and R.S. King. 2003. *Functional Assessment of the Effects of Highway Construction on Coastal North Carolina Wetlands: Comparison of Effects Before and After Construction-Phase II (Construction).* The Center for Transportation and the Environment/N.C. Department of Transportation, Report No. FHWA/NC/2002-016.

Richardson, C.J. and K. Nunnery. 2001. Ecological Functional Assessment (EFA): a New Approach to Determining Wetland Health. In Vymazal, J. (ed). *Transformation of Nutrients in Natural and Constructed Wetlands*. Backhuys Publishing (Leiden, The Netherlands). pp. 95-111.

Richardson, C.J. and K. Nunnery. 1998. Field Investigation of Wetland Functional Indicators after Highway Construction. In McComb, A.J., and J.A. Davis (eds). *Wetlands for the Future*. Gleneagles Publishing (Adelaide).

Richburg, J.A., Patterson, W.A. III, and F. Lowenstein. 2001. Effects of Road Salt and *Phragmites Australis* Invasion on the Vegetation of a Western Massachusetts Calcareous Lake-Basin Fen. *Wetlands.* 21(2): 247-255.

Richter, K.O., and A.L. Azous. 1997a. Bird Communities in Relation to Watershed Development in Azous, A.L. and R.R. Horner (eds). *Wetlands and Urbanization, Implications for the Future.* Olympia, WA.

Richter, K.O., and A.L. Azous. 1997b. Small Mammal Distribution, Abundance, and Habitat Use in Azous, A.L. and R.R. Horner (eds). *Wetlands and Urbanization, Implications for the Future.* Olympia, WA.

Richter, K.O. and Azous, A.L. 1995. Amphibian Occurrence and Wetland Characteristics in the Puget Sound Basin. *Wetlands* 15(3): 305-312.

Rosenberg, D.M., and V.H. Resh. 1993. *Freshwater Biomonitoring and Benthic Macroinvertebrates*. Chapman and Hill, New York.

Rubbo, M.J. and J.M. Kiesecker. 2005. Amphibian Breeding Distribution in an Urbanized Landscape. *Conservation Biology* 19(2): 504-511.

Salt Institute. 2001. Data on U.S. Salt Sales. Available online at: www.saltinstitute.org

Sanders, R.L. 2002. Sedimentation Rates and Metal Retention in an Urban Louisiana Swamp. Master's Thesis. Louisiana State University, Baton Rouge, LA.

Sanzo, D., and S. J. Hecnar. 2006. Effects of Road De-icing Salt (NaCl) on Larval Wood Frogs (Rana sylvatica). *Environmental Pollution.* 140(2): 247-256.

Saravanapavan, T., V. Anbumozhi, and E. Yamaji. 2004. "Using Percent Imperviousness as a Planning Tool in Watershed Management: Case Study of the Shawsheen in USA." *Journal of Rural Planning Association* 23(6).

Saravanapavan, T. 2002. Personal Communication.

Schiffer, D.M. 1989. *Effects of Highway Runoff on the Quality of Water and Bed Sediments of Two Wetlands in Central Florida*. U.S. Geological Survey Water-Resources Investigations Report 88-4200. Prepared in Cooperation with the Florida Department of Transportation.

Schueler, T. 2004. *An Integrated Framework to Restore Small Watersheds*. Center for Watershed Protection. Ellicott City, MD.

Schueler, T. R. 2003. *Snow, Road Salt, and the Chesapeake Bay*. Envirocast 1(3). Available online at: http://www.stormcenter.com/envirocast/2003-01-01/envirocast.php

Schueler, T. 2001a. The Importance of Imperviousness. In T. Schueler and H. Holland (eds). *The Practice of Watershed Protection*. Center for Watershed Protection. Ellicott City, MD

Schueler, T. 2001b. The Compaction of Urban Soils. *Watershed Protection Techniques* 3(3): 661-665.

Schueler, T. 2000. Pollutant Dynamics of Pond Muck. In T. Schueler and H. Holland (eds) *The Practice of Watershed Protection*. Center for Watershed Protection. Ellicott City, MD

Schueler, T. 1999. Adequate Treatment Volume Critical in Virginia Stormwater Wetland. In The Practice of Watershed Protection. T. Schueler and H. Holland, eds. Center for Watershed Protection. Ellicott City, MD.

Schueler, T. 1987. *Controlling Urban Runoff: a Practical Manual for Planning and Designing Urban Best Management Practices*. Metropolitan Washington Council of Governments. Washington, D.C.

Schueler, T. and K. Brown. 2004. *Urban Stream Repair Practices: Version 1.0*. Center for Watershed Protection. Ellicott City, MD.

Schueler, T. and J. Lugbill. 1990. *Performance of Current Sediment Control Measures at Maryland Construction Sites.* Metropolitan Washington Council of Governments. Silver Spring, MD.

Schueler, T. and D. Shepp. 1992. *The Quantity of Trapped Sediments in Pool Water Within Oil Grit Separators in Suburban MD.* Metropolitan Washington Council of Governments (MWCOG).

Schumm, S. 1999. Causes and Controls of Channel Incision. In Darby, S. and A. Simon (eds). *Incised River Channels—Processes, Forms, Engineering and Management*. John Wiley & Sons. New York, NY. pp. 19-34.

Searls, D.A. 1974. Influence of Vegetation of the Distribution of Small Mammals on a Waterfowl Production Area. M.S. Thesis, South Dakota St. Univ., Brookings, SD. 47 pp.

Seattle Metro, 1993. *South Base Pond Report: The Response of Wetland Plants to Stormwater Runoff From a Transit Base.* Pub. No. 775

Semlitsch, R. 1998. Biological Delineation of Terrestrial Buffer Zones for Pond-Breeding Salamanders. *Conservation Biology.* 12: 1113-1119.

Semlitsch, R. and J. Bodie. 2003. Biological Criteria for Buffer Zones around Wetlands and Riparian Habitats for Amphibians and Reptiles. *Conservation Biology*. 17(5): 1219-1228.

Available online at:
http://www.elkhornsloughctp.org/uploads/1116287022Semlitsch%20&%20Bodie%202003.pdf

Sherman, K. 1998. *Severn Sound Urban Stormwater Pollution Control Planning Report*. Ontario. Canada.

Silverman, G., M. Stenstrom and S. Fam. 1988. "Land Use Considerations in Reducing Oil and Grease in Urban Stormwater Runoff." *Journal of Environmental Systems* 18(1): 31-46.

Simons, L.H. 1985. Small Mammal Community Structure in Old Growth and Logged Riparian Habitat. In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords). *Riparian Ecosystems and their Management: Reconciling Conflicting Uses.* Gen. Tech Rep. RM-120, USDA Forest Service, Ft. Collins, CO.

Simmons, D and R. Reynolds. 1982. Effects of Urbanization on Baseflow of Selected South-Shore Streams, Long Island, NY. *Water Resources Bulletin.* 18(5): 797-805.

Sinicrope, T.L., P.G. Hone, R.S. Warren, and W.A. Nearing. 1990. Restoration of an Impounded Salt Marsh in New England. *Estuaries.* 13(1): 25-30.

South River Federation. Personal Communication with Drew Koslow, South RiverKeeper on November 29, 2006.

Spivey, M. L., and B. Ainslie. No date. Landscape Profiling and Better Wetland Restoration: How HGM Can Help. *2004 ESRI International User Conference Proceedings, Paper 1193.* Available online at: http://gis.esri.com/library/userconf/proc04/docs/pap1193.pdf

State of Minnesota Storm-Water Advisory Group (MNSWAG). 1997. *Storm-Water and Wetlands: Planning and Evaluation Guidelines for Addressing Potential Impacts of Urban Storm-Water and Snow-Melt Runoff on Wetlands.* State of Minnesota Storm-Water Advisory Group.

Steuer, J., W. Selby, N. Hornewer, and J. Pretty. 1997. *Sources of Contamination in an Urban Basin in Marquette, Michigan, and an Analysis of Concentrations, Loads, and Data Quality*. U.S. Geological Survey, Water-Resources Investigations Report 97-4242.
Sturdevant, A., C. Craft, and J. Saco. 2002. Ecological Functions of an Impounded Marsh and Three Natural Estuarine Marshes along Woodbridge River, NY/NJ Harbor. *Urban Ecosystems* 6(3): 163-181. Springer Science Business Media B.V.

Stockwell, S.S. 1985. Distribution and Abundance of Amphibians, Reptiles, and Small Mammals in Eight Types of Maine Peatland Vegetation. M.S. Thesis, University of Maine, Orono. 57 pp.

Taylor, B., K. Ludwa, and R. Horner. 1995. *Urbanization Effects on Wetland Hydrology and Water Quality*. In 3rd Puget Sound Research Meeting. Olympia, WA.

Thibault, P.A. and W.C. Zipperer. 1994. Temporal changes of wetlands within an urbanizing-agricultural landscape. *Landscape and Urban Planning* 28:245-251.

Thibodeau, F. and B. Ostro. 1981. An Economic Analysis of Wetland Preservation. *Journal of Environmental Management*, 12:19-30.

Tiner, R. W. 1984. Wetlands of the United States: Current Status and Recent Trends. US Fish and Wildlife Service. Washington, DC.

Tomson, M.B., Thibodeaux, L.J., Kan, A.T. 2003. Fate of Heavy Metals and Inorganic Compounds during Sediment Resuspension. Hazardous Substance Research Center Research. Available online at: http://www.hsrc.org/hsrc/html/ssw/tomson.pdf

Traffic Research Board (TRB). 1991. *Road Salt Use in the United States*. Unpublished report. Available online at: http://ntl/bts/gov/DOCS/HS-041_382/017-030.pdf

TRB. See Traffic Research Board.

Trimble, S. 1997. Contribution of stream channel erosion to sediment yield from an urbanizing watershed. *Science*. 278:1442-1444.
Trombulak, S.C. and Frissell, C. A. 2000. Review of Ecological Effects of Roads on Terrestrial and Aquatic Communities. *Conservation Biology*. 14(1): 18-30.

U.S. Environmental Protection Agency. 2006. *Economic Benefits of Wetlands*. The Wetland Fact Sheet Series. EPA843-F-06-004. Office of Water. Available online at: http://www.epa.gov/owow/wetlands/pdf/EconomicBenefits.pdf Accessed October 2006.

U.S. Environmental Protection Agency. 2003. Information on the Toxic Effects of Various Chemicals and Groups of Chemicals. Available online at: http://www.epa.gov/R5Super/ecology/html/toxprofiles.htm#

U.S. Environmental Protection Agency. 2002a. *Methods for Evaluating Wetland Condition: Developing an Invertebrate Index of Biological Integrity for Wetlands*. Office of Water, U.S. Environmental Protection Agency, Washington, DC. EPA-822-R-02-019. Available online at: http://www.epa.gov/waterscience/criteria/wetlands/9Invertebrate.pdf

U.S. Environmental Protection Agency. 2002b. *Methods for Evaluating Wetland Condition: Using Amphibians in Bioassessments of Wetlands.* Office of Water, U.S. Environmental Protection Agency, Washington, DC. EPA-822-R-02-022. Available online at: http://www.epa.gov/waterscience/criteria/wetlands/12Amphibians.pdf

U.S. Environmental Protection Agency. 2002c. *Methods for Evaluating Wetland Condition:Using Vegetation to Assess Environmental Conditions in Wetlands.* Office of Water, U.S. Environmental Protection Agency, Washington, DC. EPA-822-R-02-020. Available online at: http://www.epa.gov/waterscience/criteria/wetlands/10Vegetation.pdf

U.S. Environmental Protection Agenc. 2002d. *Methods for Evaluating Wetland Condition: Vegetation-based Indicators of Wetland Nutrients.* Office of Water, U.S. Environmental Protection Agency, Washington, DC. EPA-822-R-02-024. Available online at: http://www.epa.gov/waterscience/criteria/wetlands/16Indicators.pdf

U.S. Environmental Protection Agency. 1995. Values and Functions of Wetlands. The Wetland Fact Sheet Series. EPA843-F-95-001. U.S. EPA Office of Water. Washington, DC.

U.S. Environmental Protection Agency. 1993. *Natural Wetlands and Urban Stormwater: Potential Impacts and Management.* U.S. Environmental Protection Agency. Washington, DC.

USEPA. 1983. *Results of the Nationwide Urban Runoff Project: Final Report*. U.S.EPA, Office of Water, Washington, DC.

USEPA. See U.S. Environmental Protection Agency.

U.S. Fish and Wildlife Service (USFWS) and U.S. Census Bureau. 2002. *2001 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation – Quick Facts.* Washington, D.C. Available online at: http://www.census.gov/prod/2002pubs/QFBRO.pdf

USFWS. See U.S. Fish and Wildlife Service.

United States Geological Survey (USGS). 2004. Wetland Management for Amphibians in the Willamette Valley. FS 2004-3039. Available online at: http://fresc.usgs.gov/products/fs/fs-2004-3039.pdf

United States Geological Survey (USGS). 1999. *The Quality of Our Nation's Waters-Nutrients and Pesticides*. USGS Circular 225.

USGS. See United States Geological Survey.

Velinsky, D. and J.Cummins. 1994. *Distribution of Chemical Contaminants in Wild Fish Species in the Washington, D.C. Area*. Interstate Commission on the Potomac River Basin, ICPRB., Rockville, MD. Report No. 94-1.

Verry, E.S, and D.R. Timmons. 1982. Waterborne Nutrient Flow through an Upland-Peatland Watershed in Minnesota. *Ecology.* 63(5): 1456 - 1467.

Wake DB. 1991. Declining Amphibian Populations. *Science.* 253: 860.

Wang, N. and W. J. Mitsch. 1998. Estimating Phosphorus Retention of Existing and Restored Wetlands in a Tributary Watershed of the Laurentian Great Lakes in Michigan, United States of America. *Wetlands Ecological Management.* 6: 69-82.

Wardrop, D.H. and Brooks, R.P. 1998. The Occurrence and Impact of Sedimentation in Central Pennsylvania Wetlands. *Environmental Monitoring and Assessment*. 51: 119-130.

Waschbusch, R., W. Selbig, and R. Bannerman. 2000. Sources of Phosphorus in Stormwater and Street Dirt from Two Urban Residential Basins in Madison, Wisconsin, 1994-1995. *National Conference on Tools for Urban Water Resource Management and Protection.* U.S. EPA. pp. 15-55.

Watts, A.W., Ballestero, T.P., Gardner, K.H. 2006. Uptake of Polycyclic Aromatic Hydrocarbons (PAHs) in Salt Marsh Plants *Spartina alterniflora* Grown in Contaminated Sediments. *Chemosphere.* 62(8): 1253-1260.

Weller, M.W. 1981. *Freshwater Marshes*. University of Minnesota Press: Minneapolis, Minn.

Welsch, D.J., Smart, D.L., Boyer, J.N., Minkin, P., Smith, H.C., McCandless, T. L.1995. *Forested Wetlands, Functions, Benefits and the Use of Best Management Practices*. USDA Forest Service, Northeastern Area. NA-PR-01-95. Available online at: http://www.na.fs.fed.us/spfo/pubs/n_resource/wetlands/#Introduction

Wenger, W., and M. Yaggi. 2001. Environmental Impacts of Road Salt and Alternatives in the New York City Watershed. *Stormwater* 2(5).

Wenner, E.L. 1986. Benthic Macrofauna. In R.M. Devoe and D.S. Baughman (eds). *South Carolina Coastal Impoundments: Ecological Comparison, Management, Status and Use*. Vol. 2, *Technical synthesis.* South Carolina Sea Grant Consortium, Charleston, SC. pp. 255-298.

Werner, K. J. and J. B. Zedler. 2002. How Sedge Meadow Soils, Microtopography, and Vegetation Respond to Sedimentation. *Wetlands.* 22: 451-466.

Wilcox, D. A. 1986. The Effects of Deicing Salts on Water Chemistry in Pinhook Bog, Indiana. *Water Resources Bulletin.* 22(1): 57-65.

Winter, T.C. 1988. A Conceptual Framework for Assessing Cumulative Impacts on the Hydrology of Nontidal Wetlands. *Environ. Manag*. 12(5): 605-620.

Winter, T. , J. Harvey, O. L. Franke, and W. Alley. 1999. *Groundwater and Surface Water: A Single Resource*. United States Geologic Survey Circular 1139. USGS. Denver, CO.

Woo, I. and J. Zedler. 2002. *Can nutrients alone shift a sedge meadow towards dominance by the invasive* Typha x Glauca*?* Wetlands 23(3):509-521

Zedler, J.B. and Kercher, S. 2004. Causes and Consequences of Invasive Plants in Wetlands: Opportunities, Opportunists, and Outcomes. *Critical Reviews in Plant Sciences.* 23(5): 431-432.

Zeedyk, W.D. 1996. Managing Roads for Wet Meadow Ecosystem Recovery. United States Department of Agriculture, Forest Service, Southwestern Region. FHWA-FLP-96-016.

Zimmerman, K.R., Van Huttum, B., Kraak, M.H.S., and Davids, C. 1989. Trace Metals in a Littoral Foodweb: Concentrations in Organisms, Sediment and Water. *Science of the Total Environment.*87/88: 477-494.