# WILDLIFE AND CONTAMINANTS IN CONSTRUCTED WETLANDS AND STORMWATER PONDS: CURRENT STATE OF KNOWLEDGE AND PROTOCOLS FOR MONITORING CONTAMINANT LEVELS AND EFFECTS IN WILDLIFE

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## EXECUTIVE SUMMARY

For more than a decade there has been concern about the large quantities of poor quality stormwater from urban watersheds and agricultural runoff reaching and contaminating surface waters including rivers and lakes. Municipalities, public agencies and industry have been encouraged to install stormwater treatment ponds and wetlands for improving water quality under the general planning of Best Management Practices (BMPs). This method of runoff treatment is promoted as a cost effective method to divert and treat poor quality water before it enters surface waters. Some of the BMPs include ponds and constructed wetlands to remove contaminants and suspended solids primarily through sedimentation, while also creating wildlife and fish habitat.

There is concern about trace metals, excessive nutrients, agricultural pesticides, industrial by-products, roadside runoff, etc. settling in these constructed ponds and wetlands. A concentration of contaminants in the wetland may pose a toxic hazard to fish and wildlife attracted to the stormwater or wastewater management facility. However, there is relatively little information on the concentrations of contaminants in these wetlands or the specific effects of contaminated stormwater on wildlife in constructed wetlands.

In Part I of this report, Best Management Practices are examined for efficient function of constructed wetlands, and contamination occurring in constructed wetlands based on case studies, and a preliminary risk assessment to biota using the available information. Data from case studies show that persistent chemicals bioaccumulate in sediments, water and wildlife and that in some locations chemical concentrations exceed the Ontario Sediment Quality Guidelines for "low-effect" level and "severe-effect" level for aquatic animals. Therefore, populations of aquatic invertebrates, fish and benthic invertebrates will likely be impacted under some conditions. However, definitive estimates of exposure and effects ie. risk assessment, on all wildlife are simply not possible due to the lack of available information.

In Part II of this report a protocol is described for monitoring the contamination and efficiency of contaminant removal by constructed wetlands, and determining the effects on wildlife inhabiting these sites. The purpose of the protocol is to develop a reliable database for further guidance on this matter. There are three levels of monitoring recommended: baseline information, and levels I and II monitoring. Baseline data are necessary to manage and monitor all constructed wetlands. This is fundamental information such as wetland size, mean depth, basin shape, flow characteristics and a description of land use within the watershed, essential chemistry of the wastewater stream that enters and leaves and remains in the wetland, and bioassay testing of the toxicity of the water and sediments. Level 1 monitoring determines the wildlife habitat and use patterns in the wetland and collects more detailed information to define baseline chemical levels in various ecosystem compartments such as benthos and fish. These data are used in a preliminary hazard assessment to determine the need for Level II monitoring. Level II monitoring quantifies pollutant effects in wildlife inhabitating the wetland.

## RÉSUMÉ ADMINISTRATIF

Depuis plus d'une décennie, on s'inquiète de l'existence de grosses quantités d'eau pluviale de mauvaise qualité en provenance des bassins hydrographiques et du ruissellement agricole atteignant et contaminant les eaux de surface, dont les cours d'eau et les lacs. On encourage les municipalités, les organismes publics et l'industrie à créer des marais et des mares d'épuration des eaux pluviales pour faire améliorer la qualité de l'eau dans le cadre de la planification générale des meilleures pratiques de gestion (MPG). On encourage l'adoption de cette méthode de traitement du ruissellement, façon efficace de détourner et d'épurer les eaux de mauvaise qualité avant qu'elles ne pénètrent dans les eaux de surface. Parmi les MPG, citons les mares et les marais artificiels qui enlèvent les contaminants et les solides en suspension et ce, surtout grâce à la sédimentation, tout en offrant un habitat à la faune et aux poissons.

On se préoccupe des métaux à l'état de trace, des éléments nutritifs en excédent, des dérivés industriels, du ruissellement en bordure de route, etc., qui se déposent dans ces mares et ces marais artificiels. Une certaine concentration de contaminants dans les marais risque d'être toxique pour les poissons et la faune attirés vers l'eau pluviale ou le centre d'épuration des eaux usées. On dispose toutefois d'assez peu de renseignements sur la concentration des contaminants dans ces marais ou sur les effets particuliers de l'eau pluviale contaminée sur la faune dans les marais artificiels.

Dans la partie I du rapport, on examine les meilleures pratiques de gestion pour l'efficacité des marais artificiels et la contamination survenant dans les marais artificiels d'après les études de cas, ainsi que l'évaluation des risques présentés pour la biote d'après les renseignements disponibles. D'après les données des études de cas, des substances chimiques persistantes s'accumulent biologiquement dans les sédiments, l'eau et la faune et, à certains endroits, la concentration de substances chimiques dépasse celle que stipulent les lignes de conduite de l'Ontario sur la qualité des sédiments à l'égard des niveaux à l'effet le plus faible et le plus fort sur les animaux aquatiques. En conséquence, les populations d'invertébrés aquatiques, de poissons et d'invertébrés benthiques seront sans doute touchées dans certaines conditions. Du fait du manque de renseignements, il est toutefois impossible de procéder à une estimation définitive de l'exposition et des effets, c'est-à-dire à l'évaluation des risques, touchant toute la faune.

Dans la partie II du rapport, on décrit un protocole pour surveiller la contamination et l'efficacité du retrait de contaminants par des marais artificiels et pour déterminer les effets sur la faune habitant ces lieux. On préconise trois niveaux de surveillance : les renseignements de base et la surveillance de niveau I et II. Les données de base sont nécessaires pour gérer et surveiller tous les marais artificiels. Il s'agit d'une information fondamentale comme l'étendue du marais, la profondeur moyenne, la forme du bassin, les caractéristiques d'écoulement, de la description de l'utilisation des terres du bassin hydrologique, de la chimie essentielle du courant d'eaux usées qui entrent dans le marais, le quittent et y restent, ainsi que d'essais biologiques de la toxicité de l'eau et des sédiments. La surveillance de niveau I détermine l'habitat de la faune et la configuration d'utilisation des marais et recueille plus d'information détaillée pour définir le niveau des substances chimiques de base dans les divers compartiments écosystémiques comme le benthos et les poissons. Ces données servent à l'évaluation préliminaire des risques visant à établir si une surveillance de niveau II s'impose. La surveillance de niveau II quantifie les effets des polluants sur la faune habitant les marais.

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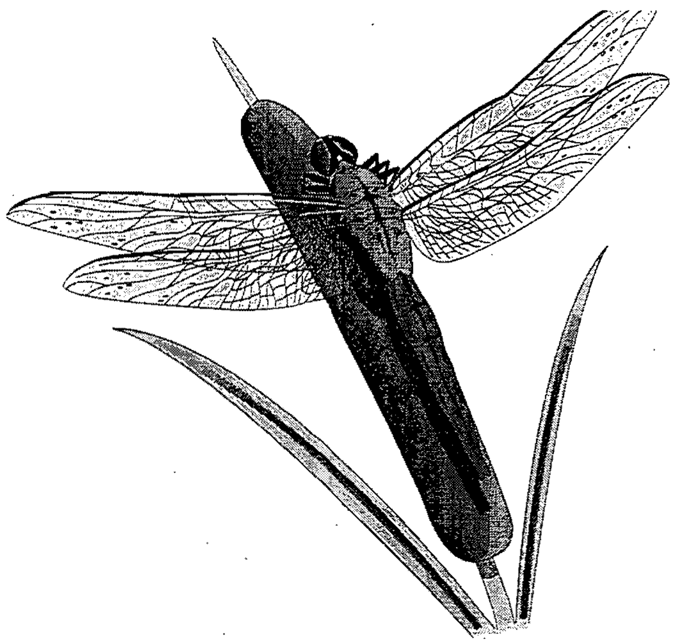
# PART I.

# STORMWATER MANAGEMENT PRACTICES USING WETLANDS. LITERATURE REVIEW OF CONTAMINATION AND EFFECTS IN WILDLIFE INHABITATING CONSTRUCTED WETLANDS AND STORMWATER PONDS



# 1. INTRODUCTION

## 1.1 Rationale and Objectives

For more than a decade there has been concern about the large quantities of poor quality stormwater from urban watersheds and agricultural runoff reaching and contaminating surface waters including rivers and lakes. Municipalities, public agencies and industry have been encouraged to install stormwater treatment ponds and wetlands for improving water quality under the general planning of Best Management Practices (BMP). This method of runoff treatment is promoted as a cost effective method to divert and treat poor quality water before it enters surface waters, while also creating wildlife and fish habitat. The purpose of these ponds and constructed wetlands is to remove contaminants and suspended solids primarily through sedimentation.

There is some concern about heavy metals such as lead and cadmium, excessive nutrients, and agricultural pesticides and herbicides settling in these constructed ponds and wetlands. The concentration of contaminants may permit bioaccumulation of some chemicals, or pose a toxic hazard to fish and wildlife attracted to the stormwater management facility. This concern was also noted in a survey of public opinion regarding the use of constructed wetlands to control stormwater contaminants (Carlisle *et al.* 1991).

There are many reviews on the accumulation of toxic contaminants in sediments, fish and wildlife in the Great Lakes (Fitchko 1986; Allan 1986; Bishop *et al.* 1992a, 1992b; Hebert *et al.* 1993). Review papers on the effects of toxic contaminants on fish and wildlife have been produced by the U.S. Fish and Wildlife Service (Eisler 1987a, 1987b, 1988, 1989, 1993). The ecological impacts of runoff from agricultural activities to surface waters have received considerable attention (e.g. Cooper 1993) and the general literature on toxicity of trace metals and chlorinated aromatic hydrocarbons is vast. The direct effects to wildlife of chemicals used in the agricultural industry have been scrutinized during the past decade and there is little doubt regarding their potential impacts to wildlife (Kendall and Akerman 1992). In fact, the practice of using mercury as a fungicide to treat seeds was reported to cause mortality of birds of prey and mammals in Sweden in the early 1950's, and represents the early link between the use of chemicals and wildlife impacts in environmental studies. However, there is relatively little information on the specific effects of contaminated stormwater on wildlife in constructed wetlands.

Marshall Macklin Monaghan (1991) found 16 out of 662 papers on constructed wetlands which dealt with fisheries and wildlife in those wetlands. In a survey on constructed wetlands in Canada, concerns with wildlife were expressed, but few respondents had information (Preiss pers. comm.). The Metropolitan Toronto and Region Conservation Authority (MTRCA) has developed a data bank of all applications for stormwater treatment facilities but while several have been constructed, little data is available and none published to date (Meek pers. comm.).

Several stormwater control tanks have been constructed (e.g. Kew Beach in Toronto and Hamilton Harbour) and the water is directed to municipal treatment plants after storm events.

In the United States, both natural and constructed wetlands have been used as a sink for municipal wastewater for many years. A survey by the U.S. Environmental Protection Agency over ten years ago reported more than 130 wetlands receiving wastewater in six midwestern states alone. Pratt and Pluto (1989) stated that "hundreds" of communities now dispose of municipal wastewater in natural and artificial wetlands. Recognizing potential problems with contaminants accumulating in wetland systems, Pratt and Pluto (ibid) asked the question, "Do wetland treatment systems become environmental hazards?". Although this important question was asked by the authors, the answer was masked under the nebulous subject of research needs. Concern for the potential harmful effects of contaminants accumulating in wetland systems has been expressed by other researchers (Bastian *et al.* 1989; Livingston 1989; Feierabend 1989). More recently, USEPA determined that information was required on the potential impacts of toxic contaminants on wildlife in wetlands (Olson 1993). This report attempts to address this issue.

## 1.2 Report Structure

**Ecological Risk Assessment**
This report is organized within the conceptual framework of an ecological risk assessment (ERA). There are four major phases within the ERA framework (Figure 1):

1. Problem formulation
2. Exposure Assessment
3. Receptor Characterization
4. Risk characterization/Hazard Assessment.

The ERA framework just described is a slight modification of the USEPA (1992) process, incorporating some terminology and useful methodology of Environment Canada (Gaudet *et al.* 1994). This document is not intended to represent a definitive hazard or risk assessment, rather, the ERA framework is used to provide a reasonably logical approach to develop the final conclusions and recommendations for this undertaking.

**PART I of this report is organized as follows:**

- An overview of the use of constructed wetlands and stormwater ponds, specifically in Ontario
- Best Management Practices for stormwater control including stormwater ponds and constructed wetlands
- Suggestions for discouraging wildlife use in constructed wetlands and stormwater ponds
- Identify chemicals of concern and efficiency of chemical removal in constructed wetlands and stormwater ponds
- Case studies of constructed wetlands
- Hazard Assessment based on available data
- Summary and Conclusions

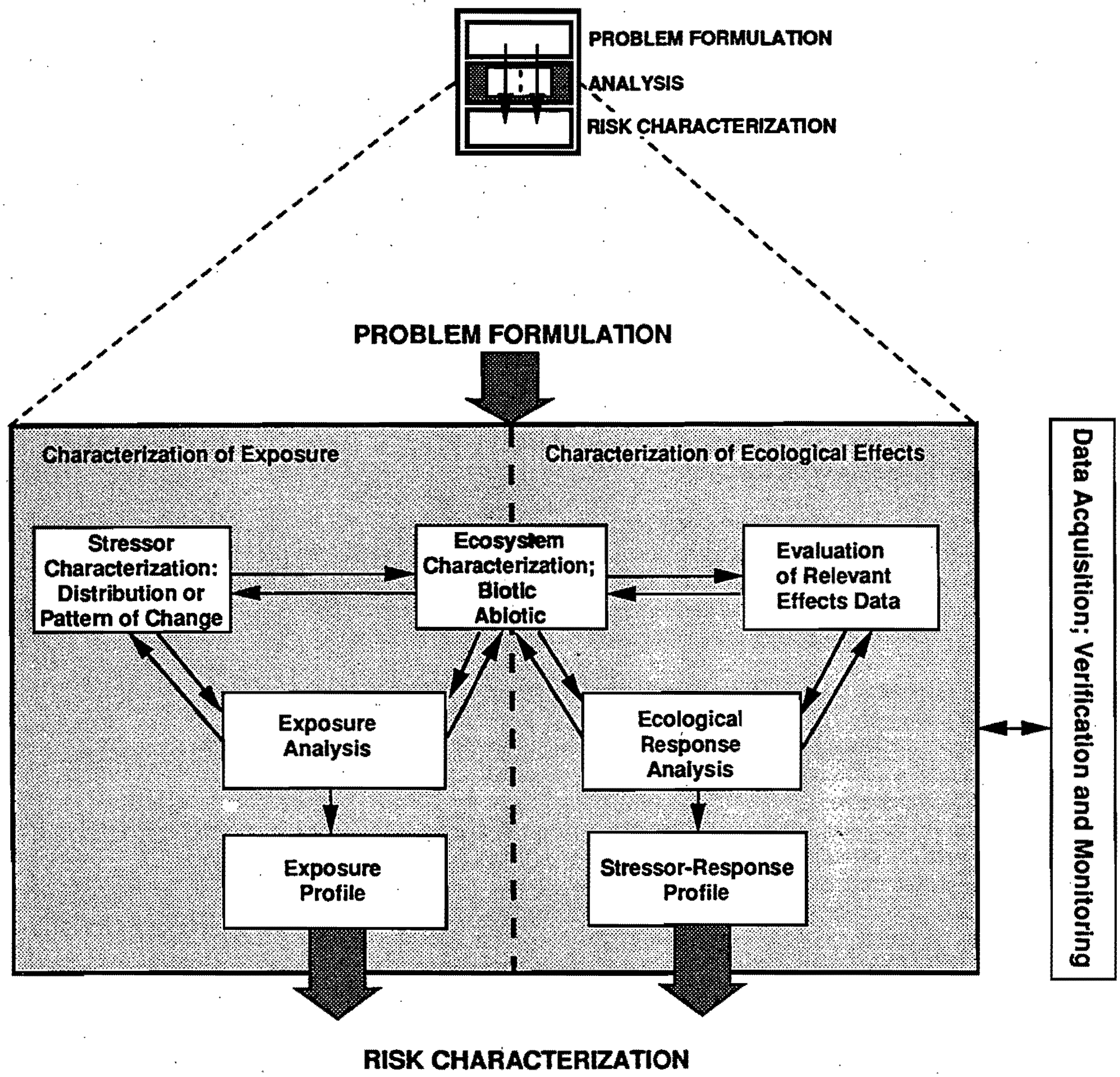



Figure 1. Approach to Ecological Risk Assessment

# 2. STORMWATER / WASTEWATER MANAGEMENT

In an urbanized area, stormwater management is an integral part of development. As natural infiltration areas (forests, soils) are replaced with impermeable structures such as roads, parking lots and buildings there is less chance for infiltration, percolation and seepage. As the proportion of impermeable area increases, direct surface runoff during storm events increases. This results in lower base flows in streams and rivers and much higher peak flows after rainfall.

Stormwater has become the major source of pollutant loading to receiving waters in many parts of North America. Best Management Practices (BMPs)(Table 1) have been developed for stormwater management (Marshall Macklin Monaghan 1991; Scheuler 1987, 1992). Vegetated systems, wet detention ponds, and wetlands are commonly used as BMPs (Hammer 1993; Olson 1993). In Florida, all newly constructed stormwater discharges must use BMPs to treat the first flush of runoff.

Many toxic substances are attached to sediment particles (Marsalek 1986) and therefore, the removal of sediments from stormwater results in the removal of associated toxic substances. While street cleaning and sediment traps in the U.S. Nationwide Urban Runoff Program (NURP) were reported to have low effectiveness for contaminant removal, wet ponds, recharge devices, grassed swales and wetlands were beneficial and improved water quality.

## 2.1 Stormwater Ponds

Stormwater ponds have become a permanent fixture in the urban landscape. For example, in the Humber River watershed there are 77 stormwater management ponds, although none of them are within Metro Toronto (MTRCA 1995).

Stormwater management ponds are designed to reduce and/or delay peak flows, and/or control the quality of runoff entering the nearest water course. Quantity ponds are designed to detain water run-off from a specific rainfall event (e.g. 2,5,10 year flood). Quality ponds are designed to delay water from a storm event for at least a 24 hour period to allow sediments and associated pollutants an opportunity to settle out before entering the closest receiving water.

It is important to distinguish between stormwater ponds and constructed wetlands. Although no clear definitions are available, stormwater ponds are primarily used to treat stormwater runoff from urban settings. The important feature is that they are designed to accommodate "peak flows" and highly fluctuating water volumes due to sudden storm events. Stormwater ponds can incorporate several features in their design including wet ponds, dry ponds, extended detention basins and any combination of these.

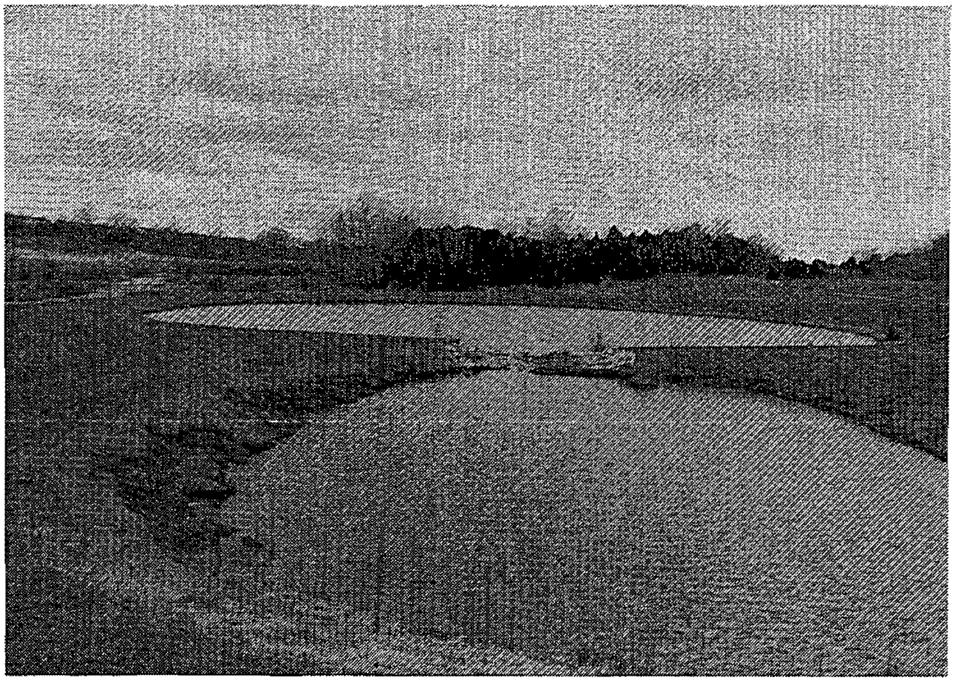


Figure 2. Stormwater ponds are now a common feature in the urban landscape

Constructed wetlands are amenable to situations where the volume of incoming water is more stable. Thus, constructed wetlands are used to treat effluent from municipal treatment plants, industrial effluent, acid mine drainage, etc. where flow is relatively constant. In some wetland situations the incoming volume can be controlled. This is necessary to maintain relatively constant water levels to support aquatic vegetation that becomes established in a wetland.

It should also be noted that BMP facilities can and often do incorporate features of both stormwater ponds and wetlands. Therefore, there is substantial overlap in the use of these terms in the literature and in everyday discussion.

**Table 1. Comparative Assessment of the Effectiveness of Urban Best Management Practices (modified from Scheuler 1992)**

| Urban BMP Options | Reliability for Pollutant Removal | Longevity | Wildlife Habitat Potential | Environmental Concerns | Special Considerations |
|---|---|---|---|---|---|
| Stormwater Wetlands | Moderate to high depending on design | 20+ years expected | High | Stream warming; natural wetland alteration | Recommended with design improvements and with the use of micropools and wetlands |
| Extended Detention Ponds | Moderate, but not always reliable | 20+ years, but frequent clogging and short detention common | Moderate | Possible stream warming and habitat destruction | Recommended with design improvements and with the use of micropools and wetlands |
| Wet Ponds | Moderate to high | 20+ years | Moderate to high | Possible stream warming, trophic shifts, habitat | Recommended with careful site evaluation |
| Multiple Pond Systems | Moderate to high, redundancy increases reliability | 20+ years | Moderate to high | Selection of appropriate option minimizes overall environmental impact | Recommended |
| Infiltration Trenches | Presumed moderate | 50% failure rate within five years | Low | Slight risk of ground water contamination | Recommended with pretreatment and geotechnical evaluation |
| Infiltration Basins | Presumed moderate, if working | 60-100% failure within five years | Low to moderate | Slight risk of ground water contamination | Not widely recommended until longevity is improved |
| Sand Filters | Moderate to high | 20+ years | Low | Minor | Recommended with local demonstration |
| Grassed Swales | Low to moderate, but unreliable | 20+ years | Low | Minor | Recommended with checkdams, as one element of a BMP system |
| Filter Strips | Unreliable in urban settings | Unknown but may be limited | Moderate if forested | Minor | Recommended as one element of a BMP system |

## 2.2 Constructed Wetlands

Constructed wetlands often consist of former terrestrial environments that have been modified to create poorly drained soils for the primary purpose of contaminant or pollutant removal from stormwater or wastewater (Hammer 1993; Marshall Macklin Monaghan 1991; Olson 1993; Scheuler 1987, 1992; Technical Practice Committee Task Force 1990). Constructed wetlands are essentially wastewater treatment systems and are designed and operated as such, though many systems do support other functional values (Hammer 1993). Designs are based on the need to retain sediments during peak flows. In this regard, the design of constructed wetlands for cropland runoff has more in common with wetlands designed for the treatment of urban runoff than those designed for treatment of secondary wastewater from municipal wastes (Baker 1993).

Proposed stormwater management wetlands usually include a sediment retention basin or pond, in which the majority of sediments can settle out. Detention time is usually between 24 and 48 captures the majority of sediments (Moss pers. comm.). It is also advisable to have several water treatment compartments or cells in a stormwater system. These normally include a sedimentation pond, constructed marshes and ponds, and meadows or woodlands for final treatment. If storm events are potentially large, then a volume storage pond (detention pond) is required at the beginning of the system. In runoff quality control practice, wet ponds are preferred to dry ponds as there is significantly better removal of contaminants. Pond geometry is important from both a functional and pollutant removal point of view (Marsalek 1990). Wetlands with sloping sides and abundant aquatic and emergent vegetation are preferred for stormwater management facilities (Marsalek 1990; Marsalek *et al.* 1992).

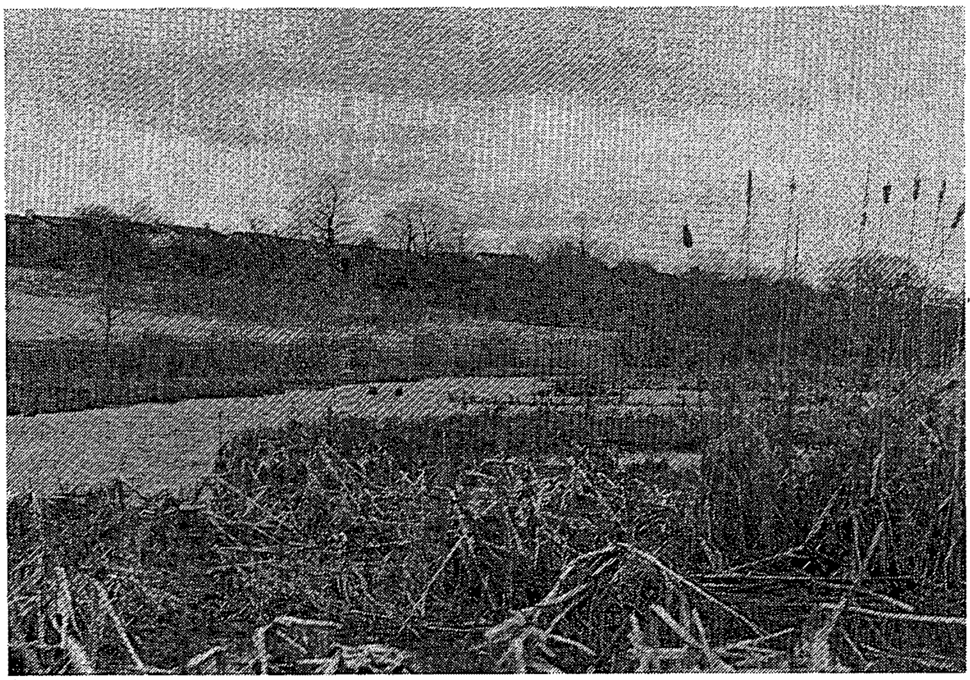


Figure 3. Constructed wetlands can improve water quality and provide wildlife habitat

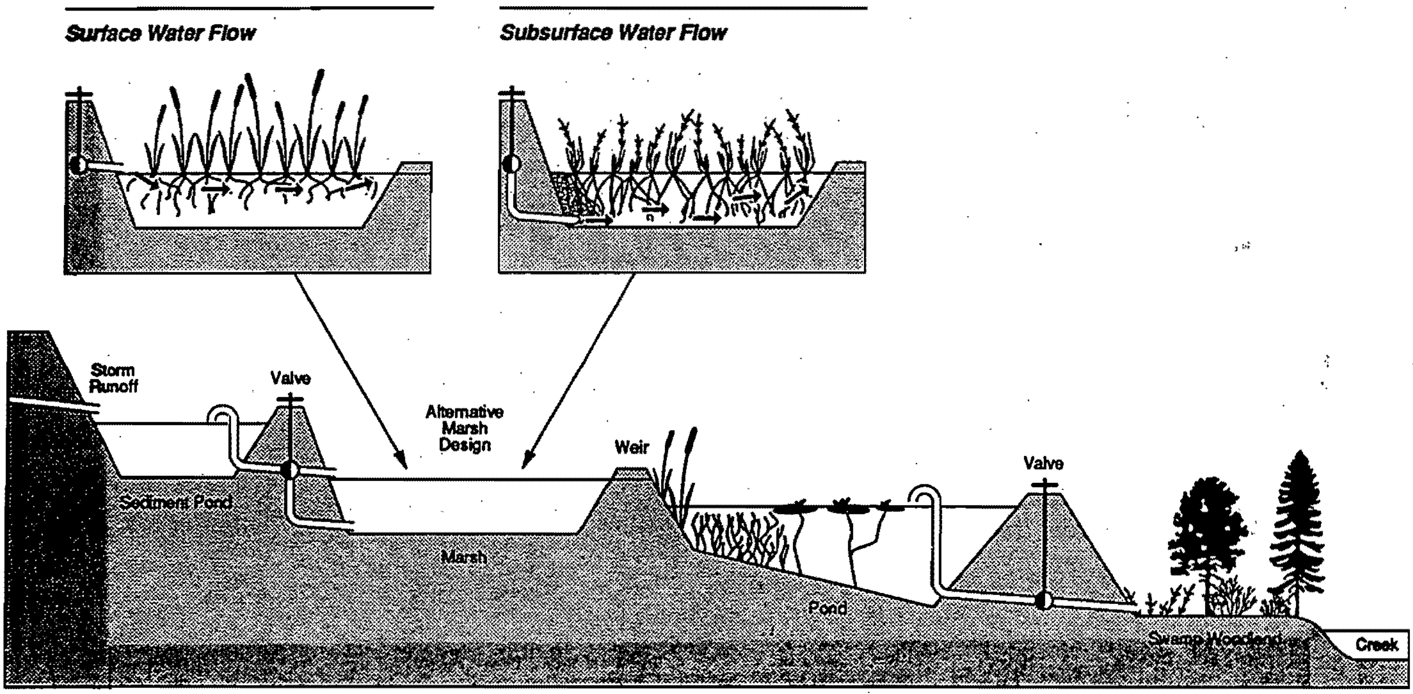



Figure 4 Schematic of Constructed Wetlands (from OMOEE and MTRCA 1992)

## 2.3 Constructed Wetlands in Ontario

A number of applications of treatment wetlands across Canada were reviewed by Pries (1994). In Ontario, the applications included treatment of municipal sewage, farm runoff, stormwater and landfill leachate. To date, monitoring of stormwater wetlands has been limited to water quality, ground water movement, and site design specifications (Litchfield and Schatz 1989; Marshall Macklin Monaghan 1991; OMOEE and MTRCA 1992; Scheuler 1987; Struger *et al.* 1994). Monitoring of vegetation, fish and wildlife communities is not required by regulating agencies and is, therefore, primarily limited to research oriented projects.

Most of the reported performance monitoring has been focused on small experimental wetlands of non-standard design (Scheuler 1992). There are six municipalities in Ontario where monitoring has been undertaken: Ottawa - Regional Municipality of Ottawa-Carleton (RMOC), Nepean - RMOC Mississauga - Lake Aquitaine and Lake Wabukayne, Richmond Hill - Mill Pond and Brampton - Heart Lake Infiltration Pipe (Marshall Macklin Monaghan 1991). Other monitoring has been undertaken of watersheds and ponds in the Hamilton and Guelph areas (Struger *et al.* 1994; Struger pers. comm.).

Existing constructed wetlands incorporate a variety of retention or detention ponds, either modifying an existing pond (Lake Wabukayne, Mississauga) or creating a wet pond (Lake Aquitaine, Mississauga, East Barrhaven, Kennedy-Burnett Pond, Bridlewood Manor Pond, Hunt Club Ridge Pond, Borden Farm Pond, Bentley Pond, Merivale Pond, Colonnade Pond and Uplands Pond)(Marshall Macklin Monaghan 1991). Two other ponds in Guelph, Fieldstone and Brandt, have been monitored for standard parameters and pesticides (Struger *et al.* 1994). None of these ponds have been systematically monitored for wildlife, and though Lake Aquitaine was stocked with rainbow trout between 1977 and 1979 there are no details available on contaminant uptake in these fish.

Although numerous municipalities in the USA apparently utilize constructed wetlands to treat municipal sewage, this is not the case in Ontario. A pilot artificial marsh project was undertaken at the Town of Listowel from 1980-1984 (Pries 1994). The results of that experiment demonstrated good removal of bacteria and nutrients by the wetland cells. Efficiency of effluent treatment was related to wetland geometry. The channelized configuration (length:width ratio of 17:1) showed better treatment efficiency than the open marsh design (length:width ratio of 4:1). The monitoring program for that pilot project did not include monitoring of potential chemical contaminants other than conventional parameters (e.g. Nitrogen, Phosphorous, Biological Oxygen Demand).

## 2.4 Other Constructed Wetlands

Approximately, 1,000 wetlands are used in North America for treating water. Some of these are wetlands constructed to treat municipal wastewaters (Kadlec and Tilton 1979; Godrey *et al.* 1985; Reed *et al.* 1987; Reddy and Smith 1987), coal and metal mine drainages (Girts and

Kleinmann 1986; Brodie *et al.* 1988; Lapakko and Eger 1988; Eger and Lapakko 1988), and wastewater produced from agriculture and textile and photography industries (Watson *et al.* 1989). The use of constructed wetlands for municipal and industrial wastewater treatment has the longest history of any application for constructed wetlands. The number of wetlands constructed for acid coal mine drainage and agricultural wastewater treatment has increased but the literature is still developing. Consequently, there is a limited database on the documentation of background levels of pesticides and metals in constructed wetlands.

Drainage from coal deposits and mining activities is a severe environmental problem that seriously degrades water quality when discharged into natural water bodies. Acid mine drainage is characterized by low pH and high iron and manganese concentrations. Pyrite ($FeS_2$) associated with coal deposits oxidizes via an initiation reaction and a propagation or catalytic reaction. These reactions can release large quantities of iron and sulphate, and the associated hydrogen ion lowers pH, which solubilizes associated metals such as manganese. Measured acid mine drainage concentrations range from 70 to 5089 u mol iron/L, 158 to 983 u mol manganese/L, 2812 to 27,270 u mol sulphate/L, and 2.7 to 6.3 standard pH units (Kleinmann and Girts 1987; Heil and Kerins 1988). Several studies (Kleinmann and Girts 1987; Brodie *et al.* 1988) have used constructed wetlands in ameliorating acid mine drainage with general consensus that they provide effective treatment in buffering acidity and reducing metal levels.

There are a variety of natural wetland treatment plants in the United States and some examples are provided by the Technical Practice Committee Task Force (1990) (Table 2). The majority of these constructed wetlands are considered successful in improving water quality, but have not been monitored with respect to contaminant uptake by wildlife.

**Table 2. Locations of Natural Treatment Systems (Technical Practice Committee Task Force 1990)**

| Type of System | Location |
|---|---|
| Slow Rate | Bakersfield California, Clayton County Georgia, Lubbock Texas, Muskegon Michigan. |
| Rapid Infiltration | Bozeman Montana, Kisimmee Florida, N. Myrtle beach South Carolina, Orlando Florida, Waycross Georgia. |
| Overland Flow | Davis California, Gordonsville Virginia, Hall's Summitt Louisiana, Heavener Florida, MacInney Florida, Raiford Florida. |
| Subsurface Flow | Mayo Peninsula Maryland, Paradise California, Otter Tail Lakes District Minnesota, Steuben Lakes Regional Waste District Indiana, Taylorsville California, Washington County Minnesota, Westboro Wisconsin. |

| Table 2. | Locations of Natural Treatment Systems (Technical Practice Committee Task Force 1990) |
|---|---|
| **Type of System** | **Location** |
| Lagoons | Corrine Utah, Eudora Kansas, Kilmichael Mississippi, Peterborough New Hampshire. |
| Constructed Wetlands | Hardin Kentucky, Incline Village Nevada, Lakeland Florida, Ocean Springs Mississippi, Orlando Florida. |
| Natural Wetlands | Cannon Beach Oregon, Hilton Head Island South Carolina, Horry county South Carolina, Houghton Lake Michigan, Walt Disney World Florida, Wildwood Florida. |
| Floating Aquatic Plants | Austin Texas, Orlando Florida, San Diego California, Sleepy Eye Minnesota, Wilton Arkansas. |

In Northern Maine, runoff from potato fields jeopardized a cold, deep lake supporting trout and salmon. The constructed wetland system consisted of a sedimentation ditch with berms leading to an overflow meadow, a cattail (*Typha* sp.) marsh, a pond, and a final polishing meadow. Results show than 80% removal of sediment, nitrogen and phosphate from rowcrop runoff. The treatment system also provides black duck breeding habitat, as well as bait fish rearing sites (cited in Hammer 1993).

From descriptions of constructed wetland systems for stormwater management a number of principles have been developed. Some of these principles for controlling nonpoint source pollution are:

- design the system for minimum maintenance
- design a system that utilises natural energies
- design the system "with" the landscape, not "against" it
- design the system with multiple objectives
- design the system as an ecotone or zone between terrestrial and aquatic ecosystems
- give the system time to develop, wetlands do not become functional overnight
- design the system for function, not form
- do not over-engineer wetland design with rectangular basins, rigid structures and regular morphology, mimic natural systems

## 2.5 Wildlife Use of Constructed Wetlands and Stormwater Ponds and Suggestions for Discouraging Wildlife Use of Contaminated Wetland Areas

One of the direct and indirect objectives of using artificial wetlands for treatment of wastewater is for creation of wildlife habitat. Natural wetlands have been systematically destroyed throughout North America to permit land development for agriculture and urban and industrial growth. It is widely accepted that over 80% of the natural wetlands in southern Ontario have been destroyed. Therefore, the construction of wetlands for wastewater and stormwater treatment is attractive to wildlife for some replacement of this vital habitat.

However, some constructed wetlands and stormwater ponds may act as primary receiving ponds in a series of wetlands designed for sewage treatment, and they may be subject to contaminant levels that are expected to be high and may be potentially dangerous to wildlife. In these cases the wetland may be designed to discourage wildlife, steps may be taken to discourage wildlife use once the wetland has been built and contaminant problems arise, or regular dredging or other contaminant removal process is necessary to avoid contamination of wildlife. If secondary and tertiary wetland cells are relatively clean and free of contaminants, it is possible to make them attractive to wildlife, and keep the contaminated areas relatively sterile with regard to cover. This will tend to keep wildlife out of the contaminated water treatment areas.

The general literature on wildlife species (both plant and animal) occurring in natural wetlands is voluminous. Feierabend (1989) outlined some of the important design features of constructed wetlands that could be incorporated to enhance them for wildlife habitat. While the use of constructed wetlands by wildlife has been documented in a few examples from the United States no example of a systematic survey of wildlife inhabitation of a constructed wetland in Ontario was found in the general literature.

Wetland and impoundment design for wildlife enhancement is relatively well known (Weller 1978, 1990). Total wildlife productivity is generally correlated with water quantity, quality and habitat features. Constructed wetlands receiving waters with high nutrient loadings generally have high wildlife populations (Knight 1993). Wildlife is attracted to wetlands that have perennial water. However, the physical design features of a wetland may have a greater influence than nutrient levels on faunal diversity and abundance (Knight 1993).

The location of wetlands in a watershed can also have an impact on its use by wildlife. If a wetland is located in the head waters (first and second order streams) it will receive only limited quantities of water between storm events (Knight 1993). As such, it will capture sediments and contaminants close to source. If, on the other hand, a wetland is located on the lower reaches (third or fourth order stream) there will be a continuous water supply but too much water may move through the wetland during storm events.

In 1979, the City of Show Low, Arizona, began disposal of some of its domestic sewage effluent into nearby Pintail Lake with additional creation of an artificial wetland complex. The project was partially funded by the U.S. EPA. Monitoring of waterfowl use of the wetland was conducted by researchers at the University of Arizona who reported a dramatic increase in nesting and waterfowl reproduction within two years (Wilhelm *et al.* 1989). Man-made islands in the wetland complex were the preferred nesting sites. The number of waterfowl nests increased from 3 in 1979, to 43 in 1980, 193 in 1981 and 380 nests in 1982. Contaminant uptake was not measured.

In the late 1970's the Amoco Oil Company incorporated a series of cascading ponds and artificial wetlands into the wastewater treatment system at its Mandan, North Dakota, refinery. The refinery is surrounded by 267 ha which are dedicated to wastewater treatment and wildlife management (Litchfield and Schatz 1989). In conjunction with state and federal wildlife groups, a substantial vegetation planting program has been undertaken including fruit bearing trees, shrubs and grasses. As a result, 191 species of birds have been observed at the facility with 51 nesting in the area. A wide number of mammals including deer (species not identified), fox (species not identified), and raccoons (*Procyon lotor*) also inhabit the area. Some of the wastewater ponds were stocked with bass, bluegill sunfish and rainbow trout. Necropsies are performed annually on rainbow trout with normal results reported to date. Tissue chemical levels were not measured in wildlife.

The Tennessee Valley Authority (TVA) constructed its first wetland to treat acid coal mine drainage in 1985. Success from that wetland lead to an accerelated program of wetland construction (Brodie *et al.* 1988). The constructed wetlands have been designed to accommodate effluent flows, limed for acid balance, and heavily planted with wetland vegetation species including cattails and wool-grass (*Scirpus cypérinus*). The downstream macroinvertebrate community responded quickly to the improved water quality. Within six months the number of invertebrate taxa increased from 2 to 19. Within the impoundment itself 32 taxa were collected six months after creation. Although only two vegetation species were planted, 20 different species were present one year after construction (Brodie *et al.* 1988). Muskrats (*Ondatra zibethicus*) were also attracted to the wetland and their burrowing caused dike failure at some locations. Dike slopes were subsequently lined with rip rap to prevent further muskrat excavation.

In Ontario, a series of four stormwater management ponds were recently designed for the City of Gloucester to clean urban runoff before entering the Rideau River. Since the ponds will be located near the Ottawa International Airport, there was concern that the ponds would attract waterfowl and gulls increasing potential hazard to aircraft.

These few examples illustrate that constructed wetlands can provide habitat for wildlife. The number and types of wildlife species using the facilities will largely depend on the system size and design. Stormwater ponds for treating urban runoff are likely to attract fewer wildlife species due their typically much smaller size.

**The following aspects of wetland design will deter wildlife from using a constructed water body:**

1. If the constructed wetlands are artificial in appearance and devoid of surrounding trees, shrubs and herbaceous vegetation, little wildlife will be attracted.

2. Shorelines should include steep banks to reduce the littoral area of shallow water. This is the feeding area most suitable for gulls, waterfowl. Deep banks would also reduce establishment of aquatic macrophytes and associate invertebrate populations. Steep slopes would also reduce ponds for nesting and resting activities as visibility will be decreased. Banks and surrounding areas should be planted with shrubs and trees. Grasses should not be cut since short grass is attractive to gulls and waterfowl.

3. It is possible to deter wildlife from using contaminated parts of wetlands. This may be done by isolating the wetland from any neighbouring watercourse and preventing fish from entering. This removes the potential for fish-eating wildlife such as kingfishers, mink (*Mustela vision*), and herons to utilize the wetland. Wildlife are generally attracted to heterogeneity in wetlands, so if a wetland is made homogenous, it lacks attractiveness for species such as ducks, muskrat and mink. Likewise, if there is no cover for wildlife around wetland cells, species such as red-winged blackbird (*Agelaius phoeniceus*) and common yellowthroat (*Geothlypis trichas*) will not have perches, or places to hide.

4. Fencing can be used to exclude larger terrestrial wildlife and is often used around sewage treatment lagoons. This may serve to keep out foxes, rabbits, or large turtles, but it will not prevent muskrat, groundhogs (*Marmota monax*), raccoons and smaller species from burrowing or climbing in. Plastic drift fencing can isolate the wetland from amphibians.

5. Passive flagging, monofilament lines, and cross fencing can be used to deter some birds, but is probably not warranted.

6. Constructed wetlands may be attractive to migratory wildlife, in particular waterfowl and shorebirds. However, if open water is not provided, ducks will find it relatively unattractive. Minimization of exposed shallow zones will decrease attraction of the area for shorebirds. Major water level fluctuations will also result in a relatively sterile shoreline.

7. In small ponds, a drift fence sourrounding the entire pond can discourage amphibian colonization of the site. However, a fence should only be used at the initial construction of the pond. Erecting a fence once the pond is dug, will stop natural movements to and from the pond by amphibians that have already colonized the site.

## 3. CONTAMINANT CONCENTRATIONS (EXPOSURE ASSESSMENT)

### 3.1 Contaminant Removal Efficiency

Low cost removal of pollutants from effluent or stormwater runoff is one of the significant goals of stormwater ponds and constructed wetlands. Several processes are involved with pollutant removal processes including:

1. filtering suspended and colloidal material from water
2. uptake of contaminants into roots and leaves of plants
3. adsorption of contaminants onto soils and plant material
4. precipitation and neutralization through generation of ammonia and bicarbonate from decay of biological material
5. precipitation of metals catalyzed by bacterial activity

An understanding of these processes is fundamental not only to designing the systems but to understanding the fate of chemicals once they have entered the pond or wetland. The relative importance of any one pathway will be a function of:

- physiochemical properties (eg. Kow, volatility) of the substance
- the hydrologic regime (retention time) of the facility
- quality of incoming water
- amount of biological matter (plants, animals) in the system

Gordine and Adams (1994) reviewed the performance of different types of stormwater ponds in Ontario including Retention Basins; Extended Detention Ponds; Sedimentation Ponds; and Wet Settling Ponds. The major design parameters for infiltration facilities include the volume of the basin, area of the bottom subject to infiltration, and hydraulic conductivity of the underlying soil. The authors suggest that shallow ponds with larger surface area are marginally more effective at removing pollutants, but that advantage may be outweighed by other considerations. They further indicate that identical ponds in different parts of Ontario may perform differently due to watershed characteristics and type of pollutants encountered.

The majority of contaminants in urban stormwater are washed out during the early stages of a rainfall event "first flush" (Ellis 1989). The quantity of contaminants in stormwater in a watershed varies considerably and depends on land use, soil type, etc. (Sator and Boyd 1972; Scheuler 1987). The U.S. EPA lists 129 priority pollutants among the many toxic contaminants in urban runoff. Calculations have been made of the annual loadings of 50 contaminants in the Ontario portion of the Great Lakes Basin and heavy metals and PAHs have received the most attention (Marsalek and Schroeter 1988).

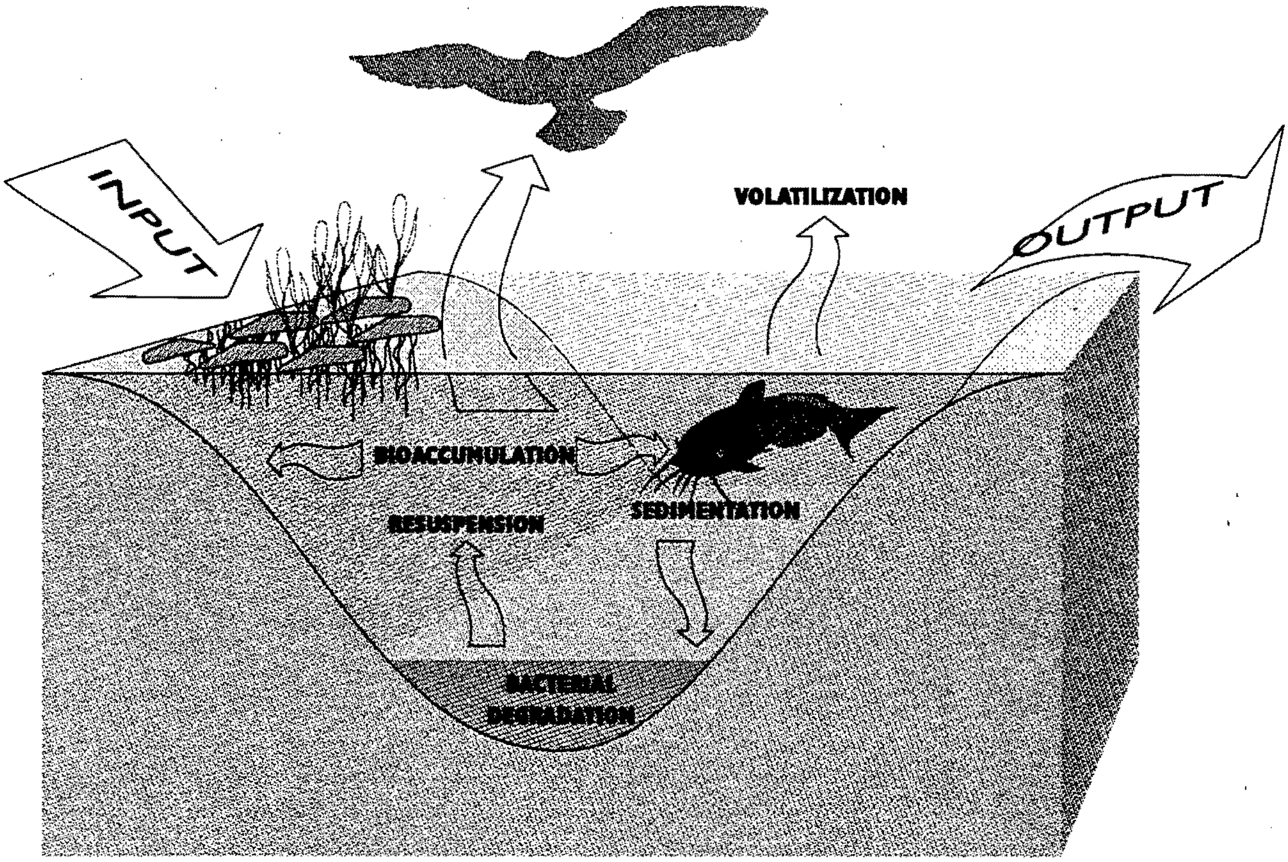



Figure 5. Chemical transport processes in constructed pond wetlands

In three urban watersheds (Windsor, Sault Ste. Marie and Sarnia) the loadings of contaminants in stormwater runoff were ranked as follows: chloride $10^6$ kg/yr, iron $10^5$ kg/yr, oil and grease $10^4$ - $10^5$ kg/yr, ammonia and phosphorus $10^4$ kg/yr, lead and zinc $10^3$ kg/yr, copper $10^2$ - $10^3$ kg/yr, nickel and phenols $10^2$ kg/yr, cyanide, PAHs, cadmium and cobalt $10^1$ - $10^2$ kg/yr, PCBs and mercury $10^0$ kg/yr, HCB $10^{-2}$ - $10^{-1}$ kg/yr (Marsalek and Ng 1987, 1989).

There are several removal pathways for pollutants in constructed stormwater wetlands (Figure 5). They include sedimentation, adsorption of contaminants to sediments/vegetation/detritus, physical filtration of runoff, microbial uptake/transformation, uptake by wetland plants, uptake by algae, and extra detention and/or retention (Scheuler 1992; Smith *et al.* 1993).

The amount of pollutant removal in wetlands varies considerably but removal values of total suspended solids (TSS) 75%, total phosphorus (TP) 45%, total nitrogen (TKN) 25%, organic carbon (OC) 15%, lead (Pb) 75%, zinc (Zn) 50% and bacteria 2 log reduction, are often obtained (Scheuler 1992). Heavy metals, pesticides and other refractory chemicals tend to become immobilized in soils or bound to organic detritus (Smith *et al.* 1993). There is also a reduction of dissolved substances in wetlands of 40-70% through plant uptake and chemical transformation in sediments and water (Martin and Miller 1987).

In Washington State, a wetland system was built to treat stormwater runoff from a subdivision. Stormwater runoff was collected into a settling tank, for removal of sediments, screened for trash and then the water fed to a 12" PVC pipe laid across a slope. Water velocity was slowed to non-erosive rates (<0.304 m/s) and let out through french drains so that flow was laminar down the slope, flowed through a vegetated swale and then entered a wetland. Water samples were analyzed for nutrients and SS and, at selected sites, for oils and grease, organophosphate pesticides, chlorinated herbicides, organochlorine pesticides, and metals (chromium, copper, lead and zinc). No herbicides or pesticides were detected in first flush samples in 1991 and 1992. Levels of chromium and zinc were detected, but were below EPA drinking water standards (Bautista and Geiger 1993).

A regional stormwater wetland system was created in Tallahassee in 1983 to reduce pollutant loads to Megginnis Arm and Lake Jackson. The 900 ha watershed includes highway, residential and commercial land uses. The stormwater treatment system incorporates an 11.5 ha detention pond, a 1.8 ha intermittent underdrain sand filter, and a 2.5 ha artificial marsh. Under normal loadings, the system removes 95% TSS, 75% TKN, 37% ammonium, 90% TP. Marsh pollutant removal is seasonal with good removal (30-60%) during growing season, and net exports during winter from plant dieback and decay (Livingston 1989).

**Table 3. Removal Efficiencies - Kennedy-Burnett (Batch Mode)(from Marshall Macklin Monaghan 1991).**

| Date | Runoff ($m^3$) | TP % | SS % | FC % | FS % | TON % | BOD % | Zn % | Pb % | Reten. (hr) |
|---|---|---|---|---|---|---|---|---|---|---|
| 1980/07/08 | 1200 | 92 | 97 | 99 | 98 | 63 | 76 | 49 | 65 | 75 |
| 1980/07/15 | 1500 | 35 | 99 | 99 | 99 | 83 | 50 | 45 | 60 | 52 |
| 1980/07/28 | 2900 | 85 | 99 | 97 | 99 | 22 | 14 | 25 | 36 | 45 |
| 1980/08/27 | 1000 | 95 | 98 | 99 | 99 | 80 | 63 | 19 | 67 | 49 |
| 1980/09/02 | 1300 | 86 | 95 | 99 | 99 | 20 | -24 | -33 | -31 | 73 |
| 1981/08/05 | 15000 | 89 | 97 | 94 | 63 | 31 | 43 | - | - | 60 |
| Average | 1580 | 79 | 98 | 99 | 99 | 54 | 36 | 21 | 39 | 59 |

## 3.2 Sediments

Sediments in stormwater is probably the most common contaminant. High levels of sediments may destroy wetlands by smothering substrates, clogging gills of aquatic organisms, and reducing light transmittance for aquatic plants. Many toxic contaminants are associated with the sediment fraction and, generally speaking, the smaller the particle size, the larger the proportion of substances adsorbed on the surface (Sator and Boyd 1972). The retention of sediments in wetlands is a major method for contaminant removal. The lower the water energy flow rates, the greater the settlement of particles.

Nightingale (1987) studied the concentration of metals in soils and sediments in five urban stormwater retention basins in Fresno, California. The ponds were built between 1962-1980 making them 1 to 19 years old at the time of sampling. The concentrations of 5 metals in the top surface (0-2 cm) sediments are reported in Table 4. The paper by Nightingale provides full soil profiles down to 60 cm, which shows that metal levels were highest in the top surface layers. The data in Table 4 indicate that the concentration of metals generally increased with age of the retention facility. Lead was the metal that accumulated the greatest relative to background levels with surface concentrations up to 1400 mg/kg. For comparison, the Ontario provincial sediment quality Severe Effect Level (SEL) for lead is 250 mg/kg.

**Table 4. Summary of Metal Levels (mg/kg) in Surface Sediments (0-2 cm) from Urban Retention Basins, Fresno, California (Nightingale 1987)**

| Parameter | Basin/Year built | | | | | |
|---|---|---|---|---|---|---|
| | F 1965 | G 1962 | M 1969 | EE 1977 | MM 1980 | Controls |
| % silt | 28 | 15 | 32 | 22 | 1 | -- |
| Iron (g/kg) | 19 | 15 | 28 | 13 | 7.7 | -- |
| Arsenic (mg/kg) | 16 | 5.9 | 29 | 5.4 | 2.0 | 0.3-12.0 |
| Nickel (mg/kg) | 27 | 36 | 40 | 22 | 6.9 | 8.6-35.0 |
| Copper (mg/kg) | 31 | 24 | 39 | 25 | 7.7 | 10-49 |
| Lead (mg/kg) | 670 | 570 | 1400 | 310 | 130 | 8.3-107 |

As part of the Hanlon Creek Watershed Study in Guelph, Ontario, levels of metals and nutrients were measured in sediments of creeks and stormwater ponds (Marshall Macklin Monaghan 1992). As would be expected, metal levels were generally higher in pond sediments than in creek sediments. The range of metal levels measured in eleven stormwater pond sediments are summarized in Table 5. The MOEE sediment quality Lowest Effect Level (LEL) was exceeded in some cases for cadmium, chromium, copper, lead, nickel, zinc, TKN and total phosphorus. The provincial SEL was not exceeded for any parameter (excluding one anomalously high nickel value of 1,100 mg/kg).

Not surprisingly, the concentration of some parameters is higher in older ponds (10-17 years) compared with ponds that were 6-8 years old at the time of sampling (Table 5). Chemicals that are elevated in the older ponds include chromium, copper, lead, zinc, ammonia, TKN, and total phosphorus. the Hanlon Creek Watershed Study concludes that while the pond sediments are "mildly contaminated", the efficiency of trapping (sediments) may not be high, given the relatively low levels of contaminants (Marshall Macklin Monaghan 1992). The inference is that the ponds were more efficient in trapping suspended sediments, contaminant levels would be higher.

Pesticide levels were measured in creek sediments, but unfortunately not in pond sediments. In all cases the pesticide concentration was below the detection limit.

| Table 5. | Hanlon Creek (Guelph) Stormwater Pond Sediment Results (Marshall Macklin Monaghan 1992) | | |
|---|---|---|---|
| **Parameter** | **Sediment Concentrations (mg/kg)** | | |
| | **Range for 11 Ponds** | **Mean for Ponds 6-8 years old (n=5)** | **Mean for Ponds 10-17 years old (n=6)** |
| Cadmium | <1-3 | <1 | 1.4 |
| Chromium | 9-27 | 12.2 | 19.0 |
| Copper | 15-68 | 27.8* | 45.7* |
| Iron (g/kg) | 78-17 | 10.8 | 13.7 |
| Nickel | 6-1100 | 8.8* | 13.4 |
| Lead | 23-200 | 39.4* | 90.0* |
| Zinc | 120-460 | 194* | 300* |
| Ammonia | <10-44 | 19.7 | 26.0 |
| Nitrite | <0.5 | <0.5 | <0.5 |
| Nitrate | <0.5 | <0.5 | <0.5 |
| TKN | 260-860 | 590 | 657 |
| Phosphorus | 420-910 | 560 | 663 |

excludes one apparently anomalous value of 1100 mg/kg
* exceeds provincial Lowest Effect Level

## 3.3 Water

### 3.3.1 Nutrients

The major nutrients which occur in wetlands are forms of nitrogen and phosphorus, which, when present in high levels, can have detrimental effects. For instance, ammonia from livestock waste may be toxic to fishes at levels as low as 0.02 mg/L, especially at high pH levels (Cooper 1993). Dairy and parlour washing and other livestock operations cause the most significant deterioration of downstream water quality but are readily rendered harmless in constructed wetlands (Maddox and Kingsley 1989). Constructed wetlands removed 91% of ammonia, 62% of organic P, and 76% of BOD when coupled with an anaerobic lagoon during the first season of operation at a dairy farm (Cooper 1993).

Johengen and LaRock (1993) studied the removal of nitrogen and phosphorous compounds from a series of constructed wetlands planted with different vegetation species. The experiments showed that mesocosms with macrophytes were about 20-30% more efficient at removing nitrogen and phosphorous than with sediments only. Direct uptake of nutrients by macrophytes may be only partially responsible for the removal, while periphyton associated with the macrophytes could account for a substantial portion of the nutrient removal.

Under aerobic conditions, nitrifying bacteria are dominant in wetlands, and ammonia is converted to nitrite by *Nitrosomonas* sp. and to nitrates by *Nitrobacter* sp. Under anaerobic conditions, denitrification is carried out by other bacteria such as *Thiobacillus denitrificans*. The oxygen status of water is important in controlling the removal of ammonia. In wetlands with good draw down, aerobic conditions are maintained and ammonia is readily broken down to nitrates.

Wetlands often become saturated with phosphates and a seasonal phosphorus flux may occur (Richardson 1985). Phosphorus is often adsorbed with sediments. Constructed wetlands were found to be more efficient (63-96%) at retaining phosphorus than natural wetlands (4-10%) in northeastern Illinois (Mitsch 1993). There does not appear to be any long term hazard to wildlife from nutrient concentrations in stormwater.

### 3.3.2 Bacteria and Viruses

Faecal coliform concentrations in urban stormwater runoff often exceed water-contact criteria at downstream swimming beaches (Marsalek *et al.* 1992). Studies have recently shown that assumed relationships between faecal coliforms and pathogenic bacteria such as *Salmonella* may not be valid. However, other pathogens such as *Pseudomonas aeruginosa* may be present in very high concentrations (Marsalek *et al.* 1992) but are not often monitored (Field and Pitt 1990).

There is a marked reduction in *E. coli* levels in many constructed wetlands particularly with adequate residence times (2-3 days). Wildlife can also contribute bacteria to wetlands. Viruses may also be reduced in numbers, but there is little data available (Gersberg *et al.* 1989).

Some micro-organisms may be injurious to wildlife (e.g. avian cholera and botulism) (Friend 1985) and parasitic nematodes (Spalding 1990). The occurrence of these microorganisms and nematodes are possible due to depressed oxygen levels but they have not been shown to be a widespread threat to the use of wetlands for priority substance control (Knight 1993). In a well constructed wetland system, removal of bacteria and viruses is high and there should be no long term hazard to wildlife.

### 3.3.3 Heavy Metals

Metals in the environment, particularly in stormwater runoff, are usually present in particulate and strongly complexed (organic) forms, with only a small fraction of the total metal concentration being bioavailable. Based on model predictions, the bioavailable fraction constituted 6% or less for total copper and lead, and about 10% to 35% for total zinc concentrations at Fresno and Salt Lake City (Paulson and Amy 1993). The short term impact of metals in stormwater runoff, based on the bioavailable fraction of total metals, is significantly less than initially concluded from the NURP studies, which were based on total concentrations (Little *et al.* 1987; Paulson and Amy 1993; Wilbur and Hunter 1980).

The partitioning of major chemicals to different size particles is summarized in Table 6.

The most common metals found in urban stormwater runoff are zinc, lead, and copper (Marsalek and Schroeter 1988). Zinc may be leached from galvanized-metal gutters and downspouts. Methyl mercury (the most biologically active form) is a minor part of the total mercury pool in the environment and is mainly a product of microbial activity. There is potential contamination of fish and wildlife from metals and salts (Eisler 1987a, 1988, 1993; Hebert *et al.* 1993).

### 3.3.4 Oils and Greases

Oils and greases enter stormwater runoff from roads, parking lots and industrial facilities. Some oils are volatile, other oils and greases are biodegradable, and others are relatively persistent and become bound to sediments. Constructed wetlands can be used to remove oils, greases and associated contaminants. Constructed wetlands consisting of a system of cascading ponds and wetlands removed oils and grease consistently during a several year monitoring period (Litchfield and Schatz 1989).

Hydrocarbon loads were observed to range from 1 mg/kg in coarse estuarine sediments to 615 mg/kg in fine sediments (Little *et al.* 1987). Under suitable conditions, microbial decomposition of motor oils and greases occurs, and waste oil may be land farmed, utilizing microbes in the soil to break down hydrocarbon chains (Seif pers. comm.).

### 3.3.5 Organics

There are thousands of anthropogenic organic compounds in the environment. Some of them cause concern because of their mutagenic, teratogenic and carcinogenic effects. For example, DDT, its DDE derivatives and dioxins bioaccumulate and have caused significant impacts on organisms (Bishop *et al.* 1992a, 1992b; Eisler 1987a, 1987b, 1988; 1989; Hebert *et al.* 1993; Giesy *et al.* 1994; Nosek *et al.* 1993).

More recently developed pesticides such as the carbamate and organophosphate insecticides generally do not have the persistence of chlorinated hydrocarbon pesticides such as DDT (Kendall and Akerman 1992, Struger *et al.* 1994) however these compounds can have severe acute toxicity to non-target invertebrates living in and around water courses.

PAHs are readily metabolized by bacteria under aerobic conditions (Clark *et al.* 1988, Portier and Palmer 1989) and organic compounds up to the size of benzopyrene can be metabolized by bacteria (Portier and Palmer 1989). These processes occur under both aerobic and anaerobic conditions and may be the best method of removing toxic chemicals from agricultural wastewaters (Portier and Palmer 1989). Despite possible impact of organic contaminants in wildlife, the accumulation of these chemicals or their effects have not been regularly measured in wildlife, sediment or water in constructed wetlands.

### 3.3.6 Miscellaneous

There are other parameters which affect the toxicity of pollutants to wildlife and can interfere with wetland functioning. These include pH, temperature, oxygen concentration, Biological Oxygen Demand ($BOD_5$), Chemical Oxygen Demand (COD), and salt balance or conductivity.

Low pH inhibits chemical and bacterial activity and changes in pH affect organic complexation to a limited degree and strongly affect adsorption processes. In the absence of competing processes, the fraction of total metals that are adsorbed is virtually nonexistent at pH levels of 5 to 6, while virtually complete adsorption occurs at pH levels of 7 to 8. As adsorption increases with pH, the remaining dissolved and bioavailable fractions of metals decrease (Paulson and Amy 1993).

The BOD in natural waters may be in the order of 0.5 to 7 mg/L (Klein 1959) while chicken wastes have a BOD of 24,000 to 67,000 mg/L (Weller and Willetts 1977). If such loadings occur they have a major impact on aquatic organisms by reducing oxygen levels as well as an indirect effect on the bioavailability of heavy metals and other contaminants in stormwaters.

High temperatures affect the toxicity of some contaminants making them more bioavailable or increasing the stress on organisms. Warm waters may attract wildlife to wetlands during winter allowing for greater exposure to contaminants.

**Table 6. Fraction of Total Constituent with each Particle Size Range (% by Weight) (from Sator and Boyd 1972)**

| Constituent | < 43µ | 43 - 246µ | > 246µ |
|---|---|---|---|
| Total Solids | 5.9 | 37.5 | 56.5 |
| $BOD_5$ | 24.3 | 32.5 | 43.2 |
| COD | 22.7 | 57.4 | 19.9 |
| Volatile Solids | 25.6 | 34.0 | 40.4 |
| Phosphates | 56.2 | 36.0 | 7.8 |
| Nitrates | 31.9 | 45.1 | 23.0 |
| Kjeldahl Nitrogen | 18.7 | 39.8 | 41.5 |
| Heavy metals (all) | 51.2 | | 48.7 |
| Pesticides (all) | 73.0 | | 17.0 |
| Polychlorinated Biphenyls | 34.0 | | 66.0 |

## 3.4 Case Studies

The performance of stormwater ponds for cleansing urban and road runoff has received considerable study during the past few years (Gordine and Adams 1994). An increasing number of actual stormwater ponds are being monitored for the efficiency of operation. Most of the monitoring programs measure water quality of the incoming water, and of the outflow water, but rarely in the pond themselves. For the purposes of this review, therefore, some extrapolation to conditions in the pond is required. The following section briefly identifies several situations where monitoring of stormwater quality is either ongoing or complete.

### 3.4.1 General Urban and Road Runoff

Attention has recently been devoted to examining water quality of road and highway runoff since road runoff has often been cited as the source of water quality impairment to surface waters. The Ontario Ministry of Transportation (MTO) has been encouraged to follow BMPs for treating highway runoff by protection agencies such as the Ontario Ministry of Natural Resources and Department of Fisheries and Oceans. The MTO has monitored the quality of highway runoff in some situations, but there is no central database of this information. In fact, recent reviews on this subject by and for the MTO relied almost solely on information from the United States with virtually no Ontario data (Lorant 1992; Thomson *et al.* 1994).

Conventional pollutants from urban runoff include suspended solids, nutrients and some metals. Marsalek (1986) studied the concentration of toxic chemicals in runoff from 12 cities in southern Ontario. Samples were analyzed for metals, total PCBs, organochlorine pesticides, PAHs and chlorinated benzenes. The paper by Marsalek (1986) reported on the frequency of detection in water and street sediment, while the actual chemical concentration data are presented in Marsalek and Schroeter (1988). Trace metals (eg. Pb, Zn, Cu) were the most prevalent with regard to the frequency of detection. Metals were also more commonly associated with sediments than water alone. Lindane was the most frequently detected pesticide, occurring in over 86% of all water samples. A comparison of the data summarized in Marselek and Schroeter (1988) suggests that the mean water concentration measured in urban runoff exceeded the Provincial Water Quality Objectives (PWQOs) for at least the following compounds; total PCBs, cadmium, copper, lead and zinc.

Sediments carried in runoff were also analyzed from the 12 urban centres, either from filtered water samples or collected directly from street particulate matter. Although these samples are not collected directly from a stormwater pond, it could be predicted that they would in fact settle out and accumulate in such basins. Mean chemical levels in these sediments exceeded the provincial sediment LEL for two PAHs; Fluoranthene and Pyrene, and the following metals: arsenic, cadmium, copper, mercury, nickel and zinc, while the sediment SEL was exceeded for lead. These results are particularly disconcerting in that the mean values exceed sediment quality criteria. Sediment samples collected directly from streets undoubtedly represent worst case scenarios, but the potential loading of these contaminants to downstream basins is obviously high.

The concentrations of various substances in stormwater runoff were reported by Marsalek and Ng (1989) for three Ontario cities (Table 7). Examination of these data reveals that provincial water quality objectives (PWQOs) are exceeded for ten parameters. Any one exceedance would indicate that freshwater biota are potentially at risk. A combination of 10 substances suggests definite risk to the health of aquatic biota in receiving environments.

Dutka *et al.* (1994) recently reported on the toxicity of bottom sediments and suspended solids collected from four stormwater ponds in the Toronto area. The ponds represent two different types of catchment basins; a) industrial: Colonel Sam Smith, Etobicoke, Tapscott Stormwater Management Pond - Scarborough, and b) residential: Unionville, Markham and Heritage Estates -Richmond Hill. Extracts from the two types of sediments were subject to a battery of toxicity tests including *Daphnia magna*, ATP-Tox, ECHA biocide monitor, Toxi-chromotest, SOS-Chromotest, Microtox, Spirillium volutans, *Panagrellus redivivus* (nematode) and lettuce seed germination and root inhibition.

Results differed between ponds, sediment type and type of bioassay (Dutka *et al.* 1994) but the authors were able to draw some general conclusions: 1) Suspended particulates which may pass uninterrupted through the pond were generally more toxic than bottom sediments, 2) *Daphnia magna,* DSTTp, SOS-chromotest and SMP appeared to be the most sensitive tests, and 3) surprisingly, sediments collected from the residential catchment basins were generally more toxic than sediments from the basins draining industrial areas.

Table 7. **Mean Concentrations of Pollutants in Stormwater in Three Ontario Cities (from Marsalek and Ng 1989)**

| Constituent | Sarnia | Sault Ste. Marie | Windsor | PWQO |
|---|---|---|---|---|
| Ammonia (mg/L) | 0.52 | 0.744 | 0.296 | no PWQO |
| Phosphorus (mg/L) | 0.3 | 0.309 | 0.231 | 0.03 |
| Chloride (mg/L) | 343 | 285 | 229 | no PWQO |
| Cadmium (mg/L) | 0.01 | 0.006 | 0.0054 | 0.2002 |
| Cobalt (mg/L) | 0 | 0.0004 | 0.0023 | 0.0004 |
| Iron (mg/L) | 5.71 | 6.96 | 5.71 | 300 |
| Lead (mg/L) | 0.23 | 0.0966 | 0.154 | 0.00025 |
| Nickel (mg/L) | 0.04 | 0.0313 | 0.0278 | 0.025 |
| Zinc (mg/L) | 0.02 | 0.0151 | 0.0033 | 0.03 |
| Cyanides (mg/L) | 0 | 0.002 | 0.003 | 0.0005 |
| Oil and Grease (mg/L) | 5.37 | 2.52 | 2.14 | no PWQO |
| Phenols (mg/L) | 0.02 | 0.0151 | 0.0033 | 0.0001 |
| Mercury ($\mu$g/L) | 0.1 | 0.029 | 0.043 | 0.2 |
| Hexachlorobenzene ($\mu$g/L) | 0.1 | 0.0005 | 0.0017 | 0.085 |
| Octachlorostyrene ($\mu$g/L) | 0 | | | no PWQO |
| PCBs ($\mu$g/L) | 0.18 | 0.0269 | 0.0888 | 0.001 |
| PAHs ($\mu$g/L) | 9.1 | 9 | 2.1 | no PWQO |

### 3.4.2 Heritage Estates, Ontario

The Ontario Ministry of Environment and Energy has conducted detailed water quality and hydrology monitoring at a stormwater management pond at a rural estate subdivision (W. Liang unpubl. data). The pond receives road and yard runoff. Water samples are collected by automated sampler during storm events and analyzed for a wide range of conventional parameters. Removal efficiency of the pond for unfiltered samples was virtually 100% for several of the metals including copper, lead, zinc and chromium. This suggests that these metals are being deposited in the ponds and will accumulate in sediments. The ponds also demonstrated good ability to filter coliform bacteria.

As with many of these monitoring programs, water quality is measured at the inflow and outlets, but not in the ponds themselves. Pond water quality must lie somewhere in between the inlet and outlet concentrations, and could be spatially variable. On at least one occasion the concentration of the following parameters exceeded the PWQO's for protection of aquatic life in inlet water: copper, lead, zinc, iron and total phosphorous.

### 3.4.3 Kingston, Ontario, Stormwater Pond

Environment Canada (Sandra Kok, Project Engineer) and Queen's University (Professor E. Watt, Dept. of Civil Engineering) are conducting a monitoring program of water quality, flow and performance of a stormwater management pond in Kingston under the auspices of the Great Lakes Cleanup Fund. The pond was constructed in 1982 and consists of two stages: a permanently wet area and a dry pond. In 1992, two constructed mini-wetlands were added to the outflow end of the pond to improve polishing efficiency. It is suggested that the stormwater pond will pretreat the runoff water by removing particulate material. The wetlands were planted with several species of local vegetation.

Preliminary results for 1991 and 1992 demonstrate that suspended solids, metals and organic chemicals (phenols and oil and grease) are filtered by the detention pond under baseflow conditions (Anderson *et al.* 1993). However, increased loading of COD and nitrogen compounds was observed under baseflow conditions.

Quality of both the inflowing and outflowing water was impaired relative to provincial standards for the protections of freshwater aquatic life. Outflow water exceeded the PWQO's for copper, phenol, nitrite, lead and zinc on occasion. Since concentrations were even higher in the inflow water, toxicity to aquatic biota in the pond may be expected.

Effective sedimentation is achieved by inducing good mixing of the influent water and uniform flow velocity distribution which favours quiescent settling. Substrate measurements showed accumulation of 15 to 20 cm of sediment over a 10 yr period, with average accretion of 2 cm/yr (Watt and Marsalek 1994). The authors suggested that removal of chemicals from the pond by biota was insignificant, although no data were provided to support this observation. Presumably this comment is based on the note that there is no significant presence of plants or other aquatic organisms in the pond. However, metal uptake by plants in other similar installations has been reported and may be important (Ellis 1989).

### 3.4.4 Ile Notre-Dame, Montreal

The City of Montreal installed a series of four constructed wetlands ("filter lake complex") in 1990 to treat water at a swimming beach in a lake constructed for Expo-67 (Vincent 1992). The wetlands occupy 20,000 $m^2$, were planted with 100,000 native aquatic plants and process 28 L/s of water. After two seasons of monitoring it was observed that the wetlands were quite efficient for reducing nutrients, but less so for faecal coliform bacteria. Nitrate levels were reduced 64 to 92%, and soluble reactive phosphorous (SRP) by 57 and 82% in the two years of monitoring. Uptake of N and P by aquatic plants was considered the primary route of removal. The removal of suspended solids was less effective with only 17 and 40% reductions observed. It was thought that bank instability and erosion was reducing the efficacy of sediment removal.

Concentrations of nitrate ($NO_3$) were relatively low (mean = 0.07 mg/L) in the inflow to begin with, and a further reduction to 0.04 mg/L in the outflow was reported. The recognized drinking water quality standard for nitrate is 10 mg/L (CCREM 1987). However, there is no nitrate water quality objective for the protection of freshwater aquatic life, since it is generally recognized that nitrate is not very toxic to fish (USEPA 1988). Coldwater salmonid species may be more susceptible to nitrate than warmwater species, but the concentrations reported to be acutely toxic to trout are generally greater than 1,000 mg/L (Nordin and Pommen 1986). Westin (1974) reported that the 96 hour $LC_{50}$ of nitrate for fingerling rainbow trout was 6,000 mg/L.

### 3.4.5 Guelph, Ontario

Detailed monitoring of water and sediment quality in two urban streams near Hamilton, and two stormwater ponds in Guelph, Ontario, has been undertaken over the past four years by Environment Canada and the University of Guelph. Samples were analyzed for metals and pesticides.

**Water Quality**

The concentrations of metals (Cd, Pb, Cu, Ni, Hg and Zn) in pond water were generally lower than Canadian and provincial water quality guidelines for the protection of freshwater biota (Licsko and Struger 1995). The one exception was zinc. The average zinc concentration in the dry pond was 51.4 ug/L, which exceeds the Ontario PWQO of 20 ug/L (revised July 1994). It is suggested that the high zinc values may be explained by the fact that roof leaders within the catchment basin of the dry pond are directly connected to the storm sewer system. In contrast, roof leaders in the wet pond basin generally drain onto lawns. Interestingly, metal levels in pond water were generally much lower than in water of two urban streams from Hamilton, Ontario. The authors suggest that the ponds are acting to cleanse the water, with subsequent deposition of metals into the sediments (see below).

Eight phenoxy herbicides were detected in the water including 2,4-D, MCPA, Mecorop, Dicamba and Picloram. Water quality guidelines are only available for a few of these chemicals. About 10% of the 2,4-D levels were above the PWQO of 4.0 ug/L, while the maximum concentration measured for 2,4-D was 14.6 ug/L. The concentrations of other phenoxy herbicides were generally below the guidelines for protection of aquatic life (Struger *et al.* 1994).

Three of eleven neutral herbicides were detected including Atrazine (the most common), Metolachlor and Trifluralin. The maximum concentration of Atrazine was 0.90 ug/L. Only three of the thirteen organophosphate pesticides analyzed for were detected. These were (avg. concentration) Chlorpyrifos, Diazinon and Dimethoate. Diazinon concentrations up to 1.04 ug/L were consistently above the PWQO of 0.08 ug/L for protection of aquatic life. The levels of Chlorpyrifos (avg. 0.148 ug/L) exceeded the PWQO of 0.001 ug/L.

**Sediment Quality**
The average concentration of four metals (Cd, Cu, Pb, Zn) exceeded the provincial sediment Lowest Effect Level. Only nickel and mercury were below sediment quality criteria (Licsko and Struger 1995). Metal levels were generally higher in sediments from the dry pond compared with the wet pond. For example, the concentration of lead in the dry and wet ponds was 203 and 88 μg/g, respectively. The concentration of zinc in the dry pond (1666 μg/g) was almost double the sediment Severe Effect Level of 820 μg/g. The elevated sediment zinc levels correspond to high zinc in the dry pond water (above).

Struger *et al.* (1994) also analyzed sediments from the two stormwater ponds for 19 organochlorine compounds. Eight chemicals were detected, with average concentrations generally below the MOEE Sediment Quality Guidelines. The one exception was p,p'-DDE which had a mean concentration of 40.4 ng/g in a wet detention pond. This exceeded the MOEE Lowest Effect Level of 5 ng/g, but was substantially less than the Severe Effect Level of 959 ng/g.

### 3.4.6 Mohawk Lake, Brantford
Mohawk Lake is a shallow man-made lake in the City of Brantford on the Mohawk Canal. The canal was originally built to enable barge traffic to bypass a 15 mile loop on the Grand River and to access the industrial core of Brantford. The lake was created in the mid 1800's by widening the canal to allow barges to turn around. Mohawk Lake was closed to commercial traffic in 1890, and used extensively for recreation in the early 1900's. However, by 1950 water quality had declined to the point that the lake was used only as an outlet for municipal storm sewers and as a sediment settling basin prior to discharge to the nearby Grand River. In addition, the lake receives leachate from several nearby old waste sites and industries.

Therefore, although not designed as such, Mohawk Lake actually serves as a large stormwater pond. Furthermore, restoration of the lake include plans for constructed wetlands to help restore water quality and provide fish and wildlife habitat (ESP 1994). A detailed analysis of water and sediment quality was undertaken in 1994 as part of a remedial action investigation (ESP 1994). Maximum water depth is about 2.5 m, while sediments have accumulated in the basin up to 2.3 m thick.

The concentration of several parameters in water exceeded the Ontario PWQOs. The maximum concentration (mg/L) of some chemicals is as follows (corresponding PWQO mg/L in brackets): phenols, 0.41 (0.001); copper, 0.010 (0.005); phosphorous, 0.05 (0.02); and zinc, 0.04 (0.02). In addition, coliform bacteria levels were high.

The sediments host a wide range of substances reflecting urban and industrial activities in the basin. Concentrations of all metals measured (Cd, Cr, Fe, Cu, Pb, Hg, Mn, Zn) except arsenic exceeded the provincial sediment LEL. Levels of lead and zinc exceeded the SEL. In addition, PAHs (polycyclic aromatic hydrocarbons) were frequently detected at low levels, although anthracene and pyrene concentrations exceeded the LEL. PCBs were also detected with two samples exceeding the LEL. A pesticide scan revealed only the presence of DDE in sediments. Although present in low amounts (0.01 to 0.03 mg/kg), these levels exceed the LEL of 0.005 mg/kg.

In summary, although not generally considered a stormwater pond, Mohawk Lake functions as a sediment retention basin. The sediment quality in particular is firm evidence that sediments accumulating in such a basin will reflect local land use activities, and provide a long lasting legacy of chemical contamination. Plans are being prepared to rehabilitate Mohawk Lake to restore some of its impaired uses. This may provide an opportunity to monitor chemical levels in abiotic and biotic components of a well established detention pond.

### 3.4.7 Prairie Potholes

There is no documented data on background levels of pesticides in "constructed" wetlands but there is information on the biological impact of pesticides in agricultural runoff on "prairie pothole" wetlands (Sheehan *et al.* 1987, 1995). The potential to impact prairie potholes is high because these wetlands are close to agricultural fields and pesticides can enter these systems directly through aerial overspray or drift or indirectly through runoff. When a "worst case" scenario is used (i.e., mixing of 100% of applied material into a shallow pond of 1m depth) data indicate that water concentrations of pyrethroid insecticides range from 1-17 ppb and for other insecticides (e.g., carbamates, organophosphates) from 17-184 ppb (see Table 8). Based on acute toxicity values for macroinvertebrates (Table 9) and the potential pesticide exposures in Table 8 all chemicals (except Carbofuran because of small data base and Dimethoate) would present moderate to high impact on this group of organisms. Since young ducks (1-7 weeks post-hatch) show a high dependence on macroinvertebrates for approximately 70% of their food resource there will be an indirect effect of low macroinvertebrate abundance on duckling growth and consequent survival. This is supported in the literature (Hunter *et al.* 1984). Decrease in abundance of macroinvertebrates following pesticide exposure will also affect fish populations since they also rely on this group for food. See Table 10 for a summary of avian and aquatic effects of two benchmark chemicals.

Recently, Tome *et al.* (1995) summarized several case studies on the effects of a commonly used herbicide (2, 4-D), several organophosphates (methyl parathion, parathion) and pyrethroids (fenvalerate, esfenvalerate) on waterfowl in prairie pothole wetlands. Results indicate decreases in aquatic invertebrate populations and mortality of waterfowl and other birds in these wetlands.

**Table 8. Insecticide Application Rates[a] and Expected Initial Concentration in Pond Water Used to Assess Exposure of Aquatic Invertebrates (from Sheehan *et al.* 1995)**

| Insecticide | Recommended Application Rate[b] (g AI $ha^{-1}$) | Expected Initial Concentration in Slough Water[c] ($\mu g\ L^{-1}$) |
|---|---|---|
| **Synthetic Pyrethroids** | | |
| cypermethrin | 28 | 3 |
| deltamethrin | 7.5 | 1 |
| fenvalerate | 97.5 | 12 |
| permethrin | 140 | 17 |
| **Other Insecticides** | | |
| azinphos methyl | 420 | 52 |
| carbaryl | 1,100 | 135 |
| carbofuran | 140 | 17 |
| chlorpyrifos | 560 | 69 |
| diazinon | 550 | 68 |
| dimethoate | 490 | 60 |
| malathion | 840 | 103 |
| methoxychlor | 1,500 | 184 |
| phosmet | 1,125 | 138 |

[a] Applicable rates are typically based on control of grasshoppers on cereal crops
[b] Recommended application rates based on those provided in insect control pamphlets for Alberta, Manitoba and Saskatchewan
[c] Concentration calculated by multiplying recommended application rate times surface area/volume ratio of the model pond

**able 9. Summary of Acute Toxicity Values for the Time Integrated Median Lethal Exposure Model Reported as Range from Available Data Sets (from Sheehan *et al.* 1995)**

| ısecticide | LC50 $mgL^{-1}$ day (number of data sets) | | | | |
|---|---|---|---|---|---|
| | Daphnia magna | Gammarus lacustris[b] | Aquatic Insects[c] | Chironomus spp. | Culex spp. |
| **ynthetic Pyrethroids** | | | | | |
| ypermethrin | 0.002 (1) | 0.00006-0.00013 (2) | 0.0006 Cd (1) | 0.0002 (1) | 0.00007 (1) |
| eltamethrin | 0.008-0.01 (2) | 0.00002p(1) | 0.000028Bp(1) | 0.00023-0.00029(2) | 0.00002-0.00019 (2) |
| envalerate | 0.0066 (1) | 0.00012-0.00014pa(3) | 0.00039-0.0037E, Pd (2) | 0.0042-0.015 (2) | 0.0040-0.0047 (2) |
| ermethrin | 0.0004 (1) | 0.0005p(1) | 0.0001 Br(1) | NA | 0.0014-0.0030 (2) |
| **ther Insecticides** | | | | | |
| zinphos ethyl | 0.0023-0.0040 (2) | 0.00056-0.0006 (3) | 0.006-0.016, Pc (3) | NA | NA |
| arbaryl | 0.01-0.014 (2) | 0.040-0.044 (3) | 0.019-0.030, Pc (3) | 0.020 (1) | 0.480 (1) |
| arbofuran | 0.020 (1) | NA | NA | NA | NA |
| hlorpyrifos | 0.016 (1) | 0.00044-0.0008 (3) | 0.036-0.050, Pc (3) | 0.0005-0.0012 (2) | 0.0012-0.0016 (3) |
| iazinon | 0.0025-0.003 (2) | 0.800-1.0 | 0.100-0.155, Pc (3) | NA | 0.024-0.031 (2) |
| imethoate | 5.000-12.800 (3) | 0.800-0.900 (3) | 0.172-0.510, Pc (3) | NA | NA |
| alathion | 0.0009-0.002 (2) | 0.0036-0.004 (3) | 0.035-0.040, Pc (3) | 0.0030-0.0074 (2) | 0.032-0.034 (5) |
| ethoxychlor | 0.0037 (1) | 0.0026-0.0047 (3) | 0.0056-0.030, Pc (3) | 0.0065 (1) | 0.0089-0.0189 (2) |
| nosmet | 0.0112 (1) | 0.003-0.008f (2) | NA | NA | NA |

NA - no appropriate data set available
Index calculated from toxicity data for *Gammarus fascitis* (f), *Gammarus pulex* (p), *Gammarus pseudolimnaeus* (ps)
Index calculated from toxicity data for *Cloeon dipterum* (Cd), *Baetis parvus* (Bp), *Baetis rhodani* (Br), *Ephemerella* sp. (E), *Pteronarcys dorsata* (Pd), *Pteronarcys californica* (Pc).

**Table 10. Summary of Expected Adverse Response of Aquatic Invertebrate Populations in Treated Ponds for Suggested Benchmark Insecticides, Carbaryl Impacts are Classified as Moderate and Permethrin Impacts as Severe (from Sheehan *et al.* 1995)**

| Insecticide | Reductions in Aquatic Invertebrates[b] | | | Time Required to Recover | General Trends | Comments | References |
|---|---|---|---|---|---|---|---|
| | Pre-treatment (control) | Post-treatment | Percent Reduction | | | | |
| Carbaryl (840 g $ha^{-1}$)[a] | <u>benthic</u><br>~ 300-1,000 ind. $m^{-2}$<br>~ 1.5 g dry wt. $m^{-2}$ | ~ 200 ind. $m^{-2}$<br>~ 0.4 g dry wt. $m^{-2}$ | ~ 30-80<br>~ 75 | little effect on chironomids; 3 weeks for ephemeropht er-ans; up to 1 year amphipods; 20 weeks for total numbers of benthic organisms; no data on recovery of open water community | elimination of sensitive populations (amphipods) for an extended period; chironomids become dominant; seasonal reductions in available invertebrate biomass | ducklings on treated ponds weighted 30% less than those on untreated ponds and had only 40% the growth rate of non-food-stressed ducklings over the initial 10 days post-treatment associated with treatment | Gibbs *et al.* (1984)<br><br>Hunter *et al.* (1984) |
| | <u>surface water</u><br>~ 60-180 ind. $m^{-3}$<br>~ 0.008 g $m^{-3}$ | ~ 20-30 ind. $m^{-3}$<br>~0.003 g $m^{-3}$ | ~ 70<br>~ 60 | | | | |
| Permethrin (35-140 g $h^{-1}$)[a] | <u>benthic</u><br>~ 2,800 ind. $m^{-2}$<br>1.4 g dry wt. $m^{-2}$ | ~ 200 ind. $m^{-2}$<br>~ 0.1 g dry wt. $m^{-2}$ | > 90<br>> 90 | oligochaetes and molluscs least affected; up to a year for many crustaceans and insects; 16 weeks for macrozoopla nk-ton; rotifers actually increased in numbers | elimination of most free-swimming and benthic arthropods; invertebrate population may remain depleted for a year or more | alterations of reproduction potential and transient effects on growth rates of fish in treated systems | Kingsbury (1876)<br><br>Kingsbury and Kreutzeizer (1979)<br><br>Kauchik *et al.* (1985) |
| | <u>open water</u><br>~ 300-1,500 ind. $m^{-3}$ | ~ 100-150 ind. $m^{-3}$ | | | | | |
| | <u>near macrophytes</u><br>~ 0.5-2.8 g dry wt. $m^{-3}$ | < 0.3 g dry wt. $m^{-3}$ | | | | | |
| | <u>zooplankton</u><br>~ 100-200 ind. $L^{-1}$ | < 1 ind. $L^{-1}$ (cladocerans, copepods) | | | | | |

[a] Treated at application rates approximately equivalent to those presently recommended for crop protection.

[b] Estimated average reduction for the initial 30 days post-treatment; biomass if unreported was estimated from taxonomic composition and abundance data from authors and mean dry wt data from Driver *et al.* (1974)

### 3.4.8 Fremont, California

A 20 ha wetlands area was created in Fremont, California, in 1983 to treat urban non-point source runoff before entering San Francisco Bay (Meiorin 1989). Degradation of general water quality in the Bay and elevated bacterial levels in nearby shellfish populations were the impetus for creation of the wetland. Use of a wetland system versus a more conventional stormwater pond was a challenge to address the problem of fluctuating flow after precipitation events. Water quality was also much more variable. As discussed earlier in this report, wetlands are somewhat more amenable to treating municipal or industrial effluent which has generally constant quantity and quality.

The wetland was divided into three separate subsystems (Figure 6) with each subsystem providing a distinct function. Systems A and B serve primarily as pretreatment to filter particles. System C was the largest single component (8.5 ha) which provided storage, detention, and a place for bacterial activity to reduce suspended and dissolved constituents. The efficacy of metal removal was somewhat dependant upon metal solubility and potential for resuspension during rainfall events. Overall removal efficiency (% removal in brackets) for several metals was as follows: lead (88%), chromium (68%), copper (31%), nickel (20%), and zinc (33%), while there was a net increase in manganese export (-110%).

Heavy metals were measured in parts of two plant species (cattail and alkali bulrush (*Scirpus robustus*)) and fish tissue (Sacramento blackfish (*Orthodon microlepidotus*) and carp). A bioaccumulation index was developed which represented the ratio of plant and animal metal levels to soil/sediment metal levels. An index (or ratio) greater than 1 indicated that bioaccumulation was occurring.

Not surprisingly, metal levels were highest in System C where the greatest amount of deposition took place. Metal uptake was much greater in cattails than in the bulrush species, with accumulation in roots most pronounced. Interestingly, metal levels in fish tissue were generally below concentrations present in the soils and sediments. However, lead, copper and zinc did accumulate in carp liver, particularly in older specimens (Meiorin 1989).

### 3.4.9 Wetlands at U.S.A. Superfund Sites

Wetlands are part of a large number of the environmental compartments affected by contamination at Superfund Sites throughout the United States. Perhaps more relevant to this narrative is the fact that constructed wetlands are being increasingly recommended as part of the remediation measures to clean up these sites. A substantive amount of information is rapidly being developed on this subject, with impact statements and benefits of constructed wetlands being documented in the Records of Decision (ROD) pertaining to particular remediation programs.

Bleiler *et al.* (1994) described how chemicals had migrated from the North Lawrence Oil Dump Site (NLODS) to a nearby wetland in New York state. High levels of lead and PCBs were detected in wetland sediments, plant and animal tissues. An ecological risk assessment determined that lead and PCB contamination of the sediments may be impacting some components of the wetland community. However, removal of the sediments would result in substantial disruption to the wetland. For example, reducing lead levels to background concentrations would require alteration to more than 50 acres of wetland.

The Record of Decision for the NLODS attempted to balance the risks associated with altering high quality habitat and risk of leaving environmental contaminants in place. By cleaning up the priority contaminated areas approximately 3.5 acres of sediments with lead in excess of 250 mg/kg (New York and Ontario "Severe Effect Level") would remain in place. More than 1.5 acres would contain in excess of 1,000 mg/kg.

In a similar situation, remediation of sediments contaminated with gallium arsenide at a New Jersey site would have totally destroyed a thriving old field wetlands (D'Alleinne and Schmitt 1994). Sediment levels substantially exceeded New Jersey Dept. of Environmental Protection and Energy (NJDEPE) clean-up criteria. Detailed evaluation and risk assessment reported there were no demonstratable ecological impacts to the wetlands of the gallium arsenic, which has different properties than elemental arsenic. Furthermore, the area was a state and federally protected wetlands meaning no development could occur, and as such exposure would be limited to the local ecosystem. Therefore, the NJDEPE agreed to a site-specific remedial objective that was 50 fold higher than existing state guidance.

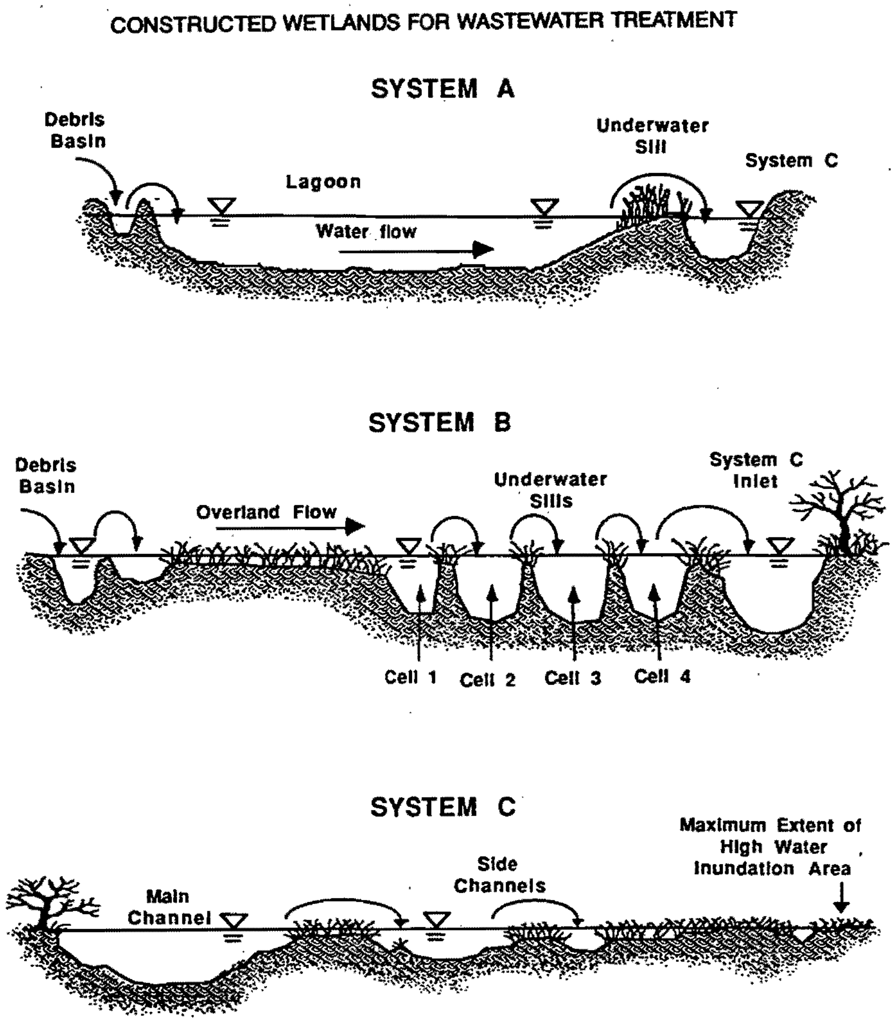



Figure 6. Schematic of Fremont wetland design (from Meiorin 1989)

# 4. HAZARD ASSESSMENT

The first important step in the hazard assessment is to compare measured exposure concentrations with existing environmental criteria. The Provincial criteria available for sediments and water are based on ecological impacts, therefore, represent a very powerful screening tool.

## 4.1 Environmental Criteria

The Ontario Sediment Quality Guidelines (Table 11) provide the "lowest-effect" level and "severe-effect" level for major contaminants in Ontario aquatic sediments. The "lowest-effect" level indicates a level of sediment contamination that can be tolerated by the majority of benthic organisms. The "severe-effect" level is the level where pronounced disturbance of the benthic community can be expected. Species used in testing are fathead minnows (*Pimephales promelas*) and burrowing mayflies (*Hexagenia limbata*) (Persaud *et al.* 1992).

Environmental criteria for water and fish tissues for protection of fish-eating wildlife are provided in Tables 12 and 13, respectively. The tables provided in this section should be used as guides to evaluate situations as new data are made available.

Data from case studies have shown sediment and water quality objectives are exceeded for several parameters in stormwater management ponds in Ontario. Therefore, populations of aquatic invertebrates, fish and benthic invertebrates will likely be impacted in these situations under some conditions. Information on constructed wetlands is not available. Furthermore, there is no information on the concentrations of chemicals in wildlife frequenting either of these habitat types (e.g. uptake), nor qualitative data on use of stormwater ponds or constructed wetlands by other wildlife groups. Therefore, definitive estimates of exposure are simply not possible on which to base a complete risk assessment for these types of wetlands. These are the reasons for the recommendations in PART II of this document describing a monitoring protocol for wildlife and abiotic contamination in wetlands built in Ontario in the future. In PART II, we provide an overview of literature on concentrations of chemicals which are toxic to wildlife. We also refer to a protocol being used by Environment Canada to develop Canadian Tissue Residue Guidelines for wildlife. Both sources of information should be used to assess the risk of health effects from chemicals to wildlife.

**Table 11. Provincial Sediment Quality Guidelines for Metals and Nutrients (values in µg/g (PPM) dry weight unless otherwise noted) (from Persaud *et al.* 1992).**

| Compound | Lowest Effect Level | Severe Effect Level |
|---|---|---|
| Arsenic | 6 | 33 |
| Cadmium | 0.6 | 10 |
| Chromium | 26 | 110 |
| Copper | 16 | 110 |
| Iron (%) | 2 | 4 |
| Lead | 31 | 250 |
| Manganese | 460 | 1100 |
| Mercury | 0.2 | 2 |
| Nickel | 16 | 75 |
| Zinc | 120 | 820 |
| TOC (%) | 1 | 10 |
| TKN | 550 | 4800 |
| TP | 600 | 2000 |
| Aldrin | 0.002 | 8 |
| BHC | 0.003 | 12 |
| Chlorodane | 0.007 | 6 |
| DDT (total) | 0.007 | 12 |
| Dieldrin | 0.002 | 91 |
| Endrin | 0.003 | 130 |
| HCB | 0.02 | 24 |
| Mirex | 0.007 | 130 |
| PCB (total) | 0.07 | 530 |
| PAH (total) | -2 | -11000 |

TOC = total organic carbon
TKN = total Kjeldahl nitrogen
TP = total phosphorus

Additional Parameters (These parameters are carried over from the Open Water Disposal Guidelines) (from Persaud *et al.* 1992)

| Compound | Concentration |
|---|---|
| Oil and Grease | 0.15% |
| Cyanide | 0.1 ppm |
| Ammonia | 100 ppm |
| Cobalt | 50 ppm |
| Silver | 0.5 ppm |

**Table 12. Summary of Ontario Provincial Water Quality Objectives and Guidelines (OMOEE 1994a) for Select Substances**

| Parameter | Concentration ($\mu$g/L) | |
|---|---|---|
| | PWQO | Revised |
| Aldrin/Dieldrin | 0.001 | |
| Aniline | 2.0 | |
| Antimony | 7.0 | |
| Arsenic | 100.0 | 5.0 |
| Benzene | 100.0 | |
| Biphenyl | 0.2 | |
| Cadmium | 0.2 | hardness 0-100 mg/L<br>PWQO = 0.1<br>hardness >100 mg/L<br>PWQO = 0.5 |
| Chlordane | 0.06 | |
| Chlorine | 2.0 | |
| Chlorobenzene | 15.0 | |
| Chrysene | 0.0001 | |
| Cineole | 100.0 | |
| Cobalt | 0.4 | |
| Copper | 5.0 | hardness 0-20 mg/L<br>PWQO = 1.0<br>hardness >20 mg/L<br>PWQO = 5.0 |
| Cyanide (free) | 5.0 | |
| Cyclohexanol | 1000.0 | |
| 2,4-D | 4.0 | |
| Dalapon | 110.0 | |
| DDT | 0.003 | |
| Diazinon | 0.08 | |
| Dibenzofuran | 0.3 | |
| Dicamba | 200.0 | |
| Dimethylamine | 3.0 | |
| Dioxane, 1,4 | 20 | |
| Diquat | 0.5 | |
| Diuron | 1.6 | |
| Dursban | 0.001 | |
| Endosulphan | 0.003 | |
| Endrin | 0.002 | |

**Table 12. Summary of Ontario Provincial Water Quality Objectives and Guidelines (OMOEE 1994a) for Select Substances**

| Parameter | Concentration ($\mu$g/L) | |
|---|---|---|
| | PWQO | Revised |
| *Escherichia coli* | 100 *E. coli*/100 ml | |
| Ethylbenzene | 8.0 | |
| Fenthion | 0.006 | |
| Heptachlor | 0.001 | |
| Hexachlorobenzene | 0.085 | |
| Hexachlorobutadiene | 0.07 | |
| Hexachloropentadiene | 0.07 | |
| Hydrogen sulphide | 6.0 | |
| Iron | 300.0 | |
| Lead<br>alkalinity (mg/L)<br><20<br>20-40<br>40-80<br>> 80 | <br><br>5<br>10<br>20<br>25 | <br><br>hardness (mg/L)<br><30 1<br>30-80 3<br>> 80 5 |
| Lindane | 0.01 | |
| Malathion | 0.1 | |
| Mercury | 0.2 | |
| Methanal | 200 | |
| Methoxychlor | 0.04 | |
| MTBE | 200 | |
| Mirex | 0.001 | |
| Molybdenum | 10.0 | |
| Naphthalene | 7.0 | |
| Nickel | 25.0 | |
| Oleic Acid | 1.0 | |
| Parathion | 0.008 | |
| Pentachlorophenol | 0.5 | |
| Phenols | 1.0 | |
| Phosphorous, total | 10-20 lakes<br>30 stream, rivers | |
| PCBs, total | 0.001 | |
| Pyrethrum | 0.01 | |
| Selenium | 100 | |

| Table 12. Summary of Ontario Provincial Water Quality Objectives and Guidelines (OMOEE 1994a) for Select Substances | | |
|---|---|---|
| **Parameter** | **Concentration ($\mu$g/L)** | |
| | **PWQO** | **Revised** |
| Silver | 0.1 | |
| Simazine | 10.0 | |
| Styrene | 4.0 | |
| Toluene | 0.8 | |
| Toxaphene | 0.008 | |
| Vandium | 7.0 | |
| Zinc | 30 | 20 |

The following are fish tissue residue criteria. As noted below, protection is provided for either human consumers of fish or the protection of fish-consuming birds. All the criteria are in $\mu$g of contaminant per gram of fish (specified as either whole fish or edible portion).

| Table 13. Table of Fish Tissue Residue Criteria (OMOEE 1994b) | |
|---|---|
| Aldrin/Dieldrin | The edible portion of fish should not exceed 0.3 $\mu$g/g for the protection of human consumers of fish. |
| DDT and Metabolites | The whole fish should not exceed 1 $\mu$g/g (wet weight basis) for the protection of fish-consuming birds. As well, the edible portion of fish should not exceed 5000 $\mu$g/g for unrestricted consumption by humans. |
| Dioxins and Furans | The edible portion of fish should not exceed 15 ng/g for unrestricted consumption by humans. This value is based on the total toxic equivalency of 2,3,7,8-TCDD and covers 15 different isomers of dioxin and furan. |
| Endrin | The edible portion of fish should not exceed 0.3 $\mu$g/g for the protection of human consumers of fish. |
| Heptachlor and Heptachlor epoxide | The edible portion of fish should not exceed 0.3 $\mu$g/g for the protection of human consumers of fish. |
| Lindane | The edible portion of fish should not exceed 0.3 $\mu$g/g for the protection of human consumers of fish. |
| Mercury | The whole fish should not exceed 0.5 $\mu$g/g (wet weight basis) for the protection of aquatic life and fish-consuming birds. As well, the edible portion of fish should not exceed 0.5 $\mu$g/g for unrestricted consumption by humans. |
| Mirex | The edible portion of fish should not exceed 100 $\mu$g/g for unrestricted consumption by humans. |
| PCBs | The edible portion of fish should not exceed 2000 $\mu$g/g for unrestricted consumption by humans. |

## 5. SUMMARY AND CONCLUSIONS

1. Constructed wetlands and stormwater ponds are effective at trapping contaminants from stormwater in urban and agricultural environments.

2. There is very little information available on the uptake of contaminants by wildlife in constructed wetlands, but elevated levels of metals in fish tissue have been documented.

3. There are numerous examples of stormwater management ponds in Ontario, with elevated levels of several metals and pesticides observed in runoff and sediments.

4. Populations of invertebrates or fish present in stormwater ponds are likely affected by reduced water quality and sediment loading as indicated by exceedances of water and sediment criteria.

5. Stormwater ponds have lower potential as wildlife habitat than constructed wetlands for vertebrate classes such as birds, fish and mammals but actual use by these groups has not been investigated.

6. Regular maintenance of stormwater ponds, e.g. dredging contaminated sediments, may reduce their potential as sources of chemical exposure to wildlife.

7. There are an increasing number of constructed wetlands in Ontario but monitoring of contaminants within the constructed wetlands or in wildlife is not routine. Studies from the United States suggest wildlife will be attracted to artificial wetlands, and potentially exposed to contaminants that accumulate in these systems.

8. Monitoring of contamination in water and sediment and exposure of wildlife to contamination in stormwater ponds and constructed wetlands is necessary. In Part II of this report a three step approach is recommended for evaluating contamination in these wetlands.

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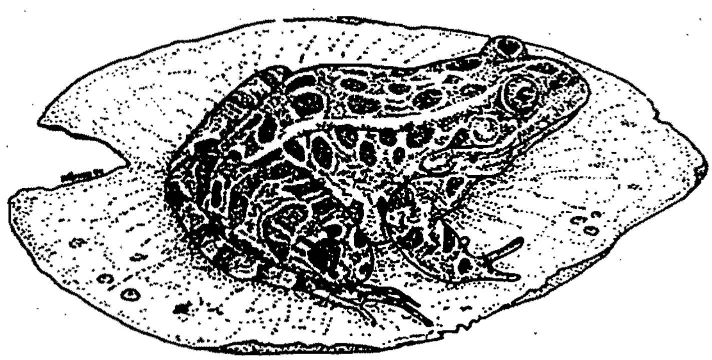
Protocols

# PART II.

# MONITORING PROTOCOLS FOR CONTAMINANT LEVELS AND EFFECTS IN WILDLIFE INHABITING CONSTRUCTED WETLANDS AND STORMWATER PONDS



# 1. INTRODUCTION AND RATIONALE

## 1.1 Background

Prior to developing this monitoring protocol Environment Canada conducted a literature review to determine the extent of available information on chemical accumulation in constructed wetlands used for effluent treatment (PART I, this report). The review process revealed that the use of constructed wetlands for effluent treatment is a rapidly emerging field, particularly in Canada, with considerable interest and proposed applications of this technology. Wetlands are effective in removing conventional pollutants such as suspended solids, excessive nutrients, metals and biochemical oxygen demand. However, there is very little information on the extent of contamination that might be occurring in these wetlands and almost no information on the contamination and effects in wildlife feeding in these sites. Nevertheless the information that is available indicates the contaminant accumulation can and does occur in wildlife inhabiting constructed wetlands and stormwater ponds (Part I, this report).

Constructed wetlands normally consist of former terrestrial environments that have been modified to create poorly drained soils for the primary purpose of contaminant or pollutant removal from stormwater or wastewater (Hammer 1993; Olson 1993; Scheuler 1987, 1992; Technical Practice Committee Task Force 1990). Constructed wetlands are essentially wastewater treatment systems and are designed and operated as such, though many systems do support other functional values (Hammer 1993).

At present, the Best Management Practices recommended for urban wetland treatment facilities are a series of ponds which usually includes an initial retention basin for coarse sediments, followed by compartments or cells such as constructed marshes and ponds, and meadows or woodlands for final treatment. If storm events are potentially large and/or frequent, then a volume storage pond (detention pond) is required at the beginning of the system.

Several processes are involved with pollutant removal processes (Scheuler 1992; Smith *et al.* 1993) in constructed wetlands and stormwater ponds including:

- filtering suspended and colloidal material from water
- uptake of contaminants into roots and leaves of plants
- adsorption of contaminants onto soils and plant material
- precipitation and neutralization through generation of ammonia and bicarbonate from decay of biological material
- precipitation of metals

An understanding of these processes is fundamental not only to designing the systems but to understanding the fate of chemicals once they have entered the pond or wetland. The relative importance of any one pathway will be a function of:

- physiochemical properties (eg. Kow, volatility) of the substance
- the hydrologic regime (retention time) of the facility
- quality of incoming water
- amount of biological matter (plants, animals) in the system

The Ontario Ministry of the Environment and Energy (OMOEE) has developed Provincial Water Quality Objectives (PWQOs) and Sediment Quality Guidelines (SQGs) that can be used as benchmarks for evaluating environmental monitoring data. The guidelines are useful as a first screening tool in the hazard assessment process for a constructed wetland; if the guidelines are exceeded, further investigation of the situation is necessary. Also, even when the Provincial guidelines for water and sediment are not exceeded but an indication of toxicity is found such as dead or diseased wildlife, sublethal but toxic responses in sensitive organisms, or a change in diversity of species occurring at the site then further investigation of the wetland is certainly warranted (see Level I and II monitoring in this report).

The question of potential hazard of chemical buildup to wildlife has been asked in numerous papers and articles on the subject (Friend 1985; Bastian *et al.* 1989; Livingston 1989; Pratt and Pluto 1989; Carlisle *et al.* 1991) but very little actual effort seems to have been directed at addressing the issue. For example, Piest and Sowls (1985) stated that contaminant levels were low in ducks using a sewage marsh in Arizona (although no data were provided) and apparently posed no hazard to wildlife. The authors did acknowledge, however, that mortality of birds due to contaminant exposure has occurred in birds using wetlands receiving sewage (Nero 1964) and that research is needed to evaluate the threat of contaminants in wetlands to wildlife. It is well established that wildlife such as waterfowl using sewage facilities for habitat can accumulate substantial levels of organic contaminants (Gebauer and Weseloh 1993; Custer *et al.* 1996).

Landers and Knuth (1991) reviewed seven USEPA funded artificial wetlands designed for water quality improvement. They noted that most projects lacked adequate monitoring to determine adverse ecological effects. Hershberger *et al.* (1995) state that low-cost wetland treatment facilities could theoretically become contaminated lagoons, and that wildlife habitat could be severely impacted.

The contaminant monitoring protocol described here was developed to provide guidance to proponents of constructed wetlands for effluent treatment. The intent of the protocol is to ***outline*** the ***basic*** components of a monitoring program. Once the basic monitoring requirements are incorporated into a monitoring program, site-specific studies may be undertaken on a case by case basis.

**Part II of this report is organized as follows:**

- Types of Constructed Wetlands and Applications
- Baseline Monitoring
- Level I Monitoring
- Data Evaluation: Hazard Assessment
- Level II Monitoring

## 2. TYPES OF CONSTRUCTED WETLANDS AND APPLICATIONS

There are essentially two basic types of constructed wetlands: Surface Flow (SF), sometimes also referred to as free-water surface, and Subsurface Flow (SSF) sometimes referred to as a submerged-bed type. The SF type of wetland (Figure 1) has an open water surface, with emergent vegetation planted in the soil/sediments. A natural or synthetic liner may be used depending on the required flow characteristics and porosity of the underlying medium. A variation on this type of facility uses free-floating aquatic plants (e.g. duckweed (*Lemna* sp.) and water lilies (*Nymphea* sp., *Nuphar* sp.)) in place of, or in conjunction with rooted plants.

In the subsurface flow wetland there is no open water. The facility is designed to maintain a saturated soil that will support emergent vegetation similar to that in a Surface Flow wetland, but without open water. The inlet pipes entering a wetland may be above the water or soil surface or discharge directly into the water. Often, constructed wetlands contain a number of cells with flow going directly from one cell into the next.

Constructed wetlands can take on any number of shapes (Figure 2) either to simulate a natural feature, or to conform to a simple geometrical design. Square or rectangular shapes are often used when several cells are linked together. These shapes facilitate even water flow through the facility and permit easier calculation of hydraulic characteristics for retention times and loading rates. Irregularly shaped basins are more likely to mimic a natural wetland if creation of wildlife habitat is an important consideration. Irregularly shaped basins will tend to require more space to treat an equivalent amount of wastewater compared with a simple configuration.

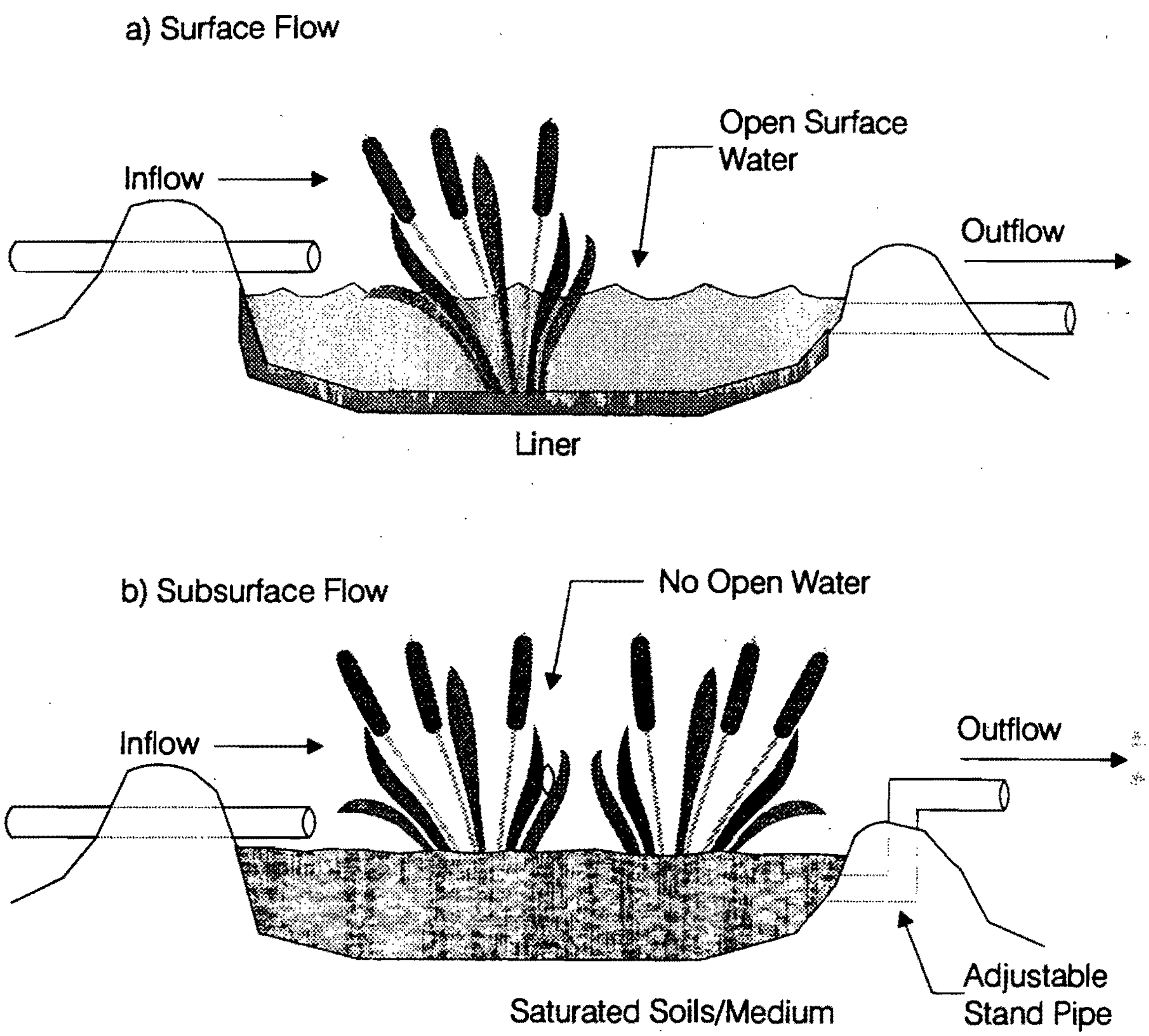



Figure 1. Illustration of the Two Basic Types of Constructed Wetlands Surface Flow (SF) and Subsurface Flow (SSF)

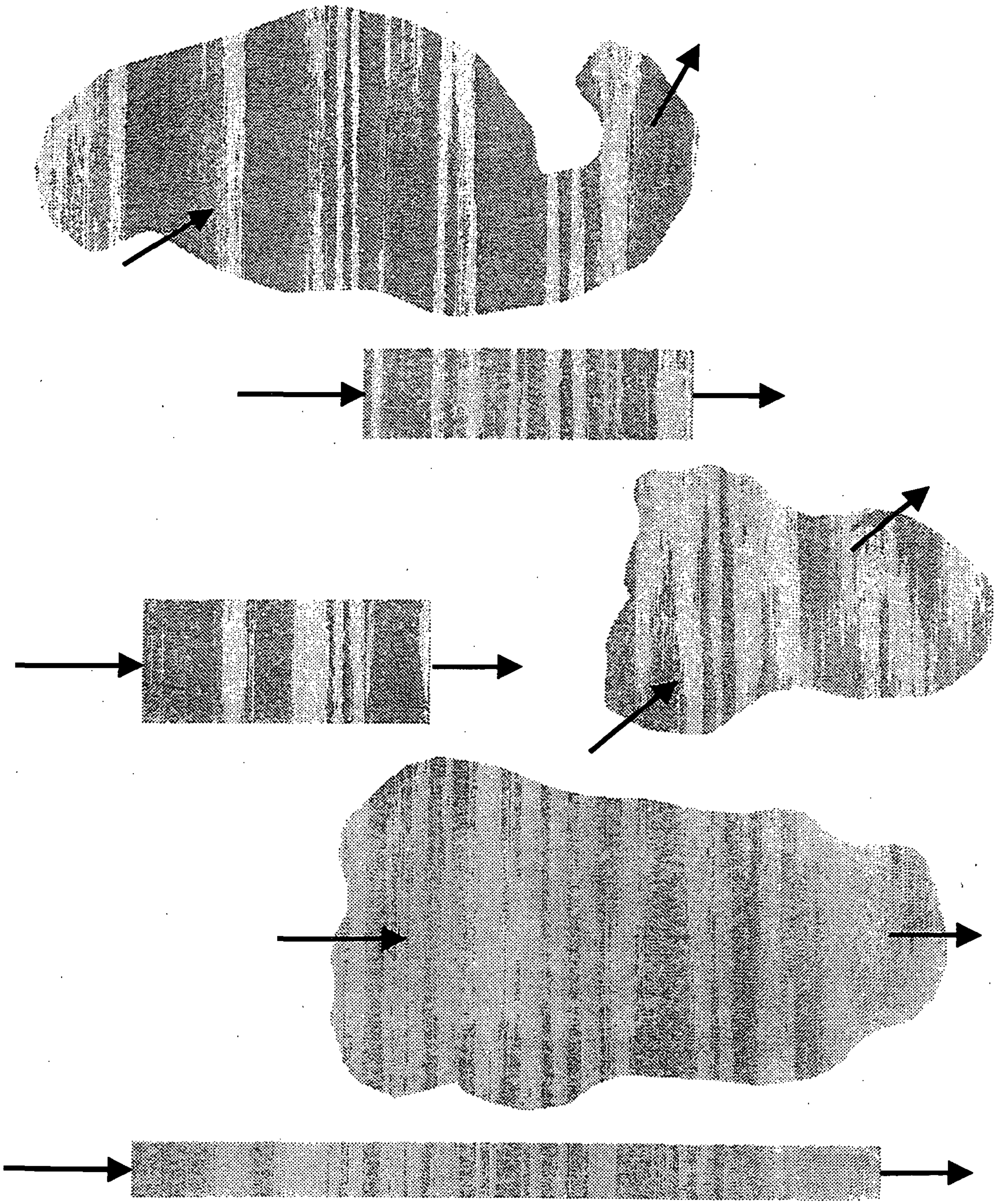


Figure 2.        Possible Basin Shapes for Constructed Wetlands

## 3. OVERVIEW OF MONITORING REQUIREMENTS

The suggested monitoring program for constructed wetlands is designed as a sequential series of steps. Each step leads to more detailed investigation, if ecological impacts are shown or suspected.

The first component of the monitoring program is to document environmental baseline conditions. These are described in more detail in Section 4, and include fundamental information on wetland size, mean depth, basin shape, flow characteristics and a description of land use within the watershed. Also included is the essential chemistry of the wastewater stream that is being treated. This type of information is normally required to design the wetland for effluent or runoff treatment and should be readily available. It is useful to have the information summarized in one concise report.

**Baseline data requirements include:**

- wetland size (surface area)
- water depth
- inflow volume
- expected retention time
- inflow/outflow water quality ; 3-4 times per year
- sediment contamination- once per year
- toxicity testing; at minimum every five years

Level 1 monitoring includes collection of more detailed information to define baseline chemical levels in various ecosystem compartments and contaminant pathways. These data are used in a preliminary hazard assessment to determine the need for Level II monitoring.

**Level I monitoring includes:**

- continuation of Baseline Monitoring
- vegetation/ wildlife habitat evaluation
- receptor identification i.e. identify benthic community; wildlife use
- if warranted, contaminant analysis of benthos; fish

The data collected during the Baseline and Level I monitoring phase should be carefully screened to determine if there is a potential for adverse affects to wildlife, including aquatic biota. The first step compares chemical levels in water and sediment to appropriate criteria. These may include Provincial Water Quality Objectives (PWQOs) as a first screening. However, if the wetland is classified as a waste treatment facility, the PWQOs are not appropriate within the system, and other effluent regulations as established by the Ontario Ministry of the Environment and Energy should be consulted.

Accumulation of chemicals in biota will indicate the presence of potentially harmful chemicals in the food chain and will also suggest the need for further monitoring. However, not all chemicals bioaccumulate yet they may have toxic effects.

Qualitative assessment of the facility to provide wildlife habitat is an important step during Levels I and II. Considerations for assessing wildlife habitat are provided in this report. The potential to induce adverse effects is dependent on the presence of a sensitive receptor. Small wetlands with very little available habitat may not support wildlife, therefore, there is little risk. However, more extensive wetlands that have been partially designed to encourage aquatic vegetation will attract waterfowl or other animals.

Level I monitoring should be repeated every 2-3 years. The decision regarding further monitoring should be made by a multidisciplinary team including the wetland operator and/or designer, and personnel with expertise in wildlife ecology, wetland vegetation and environmental chemistry and toxicology. The team may feel that certain conditions exist, either within the wetland or immediately downstream, that pose potential hazard to certain biota. However, relatively simple design or operating parameters might be modified to reduce or eliminate the problem. If these mitigation measures are undertaken, Level I monitoring can be repeated to determine whether the measures were successful in addressing the issue. If mitigation is not feasible or successful, Level II monitoring is required to determine the health effects in wildlife.

Level II monitoring includes more detailed assessment to measure and quantify health effects in wildlife due to contaminant exposure. An important consideration at the beginning of Level II is to clearly identify potential species or groups at risk (plants, animals) to focus the subsequent monitoring program. This step could include:

- intensive exposure monitoring in abiotic media and if warranted, biota;
- examination of health effects in wildlife

There are a wide range of monitoring techniques available for Level II assessment. Many of them require trained personnel with expertise in a particular discipline. This document provides a brief introduction to the methods available which will help practitioners select monitoring components suitable to the situation. However, further information and guidance will be required to design and implement an effective program.

## 4. BASELINE DATA REQUIREMENTS

An understanding of fundamental hydrology and water quality of the wetland is important to document environmental baseline conditions. Constructed wetlands may be used to treat urban or rural non-point source runoff. In these instances, the characteristics of the watershed being drained will govern the quantity and quality of runoff. The quantity of flow into the treatment wetland will be a function of drainage basin size, precipitation runoff coefficient for the watershed and land use within the basin. Therefore, it is necessary to know basic parameters such as:

- wetland size (surface area)
- water depth
- inflow volume
- expected retention time
- inflow/outflow water quality ; 3-4 times per year
- sediment contamination- once per year
- toxicity testing; at minimum every five years

The baseline report should also include photographs of the wetland from different perspectives. These will illustrate characteristics and vegetation growth that may be difficult to quantify, but will assist qualified personnel to make some judgement on the potential of the particular wetland to function as wildlife habitat.

If a wetland is built for the purpose of effluent treatment, the total catchment area is likely just the area enclosed by the containing berms and any surrounding roads. For small and medium size wetlands, the catchment area will typically be about 25% of the wetland surface area. An important characteristic of wetlands used for effluent treatment is that the inflow volume is relatively constant. With an outlet water control structure, there is little or no variation in water level within the wetland. Therefore, only vegetation able to withstand being continuously submerged will survive.

Figure 3 illustrates the basic components of a water budget for a constructed wetland. It is expected that the designers of these facilities will be able to provide most of the basic physical dimensions.

### 4.1 Hydrology and Flow

The procedure for flow measurement will depend on the type of structure used to convey water into and out of the wetland and the degree of accuracy required for the hydraulic balance.

Accurate flow measurements can be obtained when flow is directed through a weir or flume of known hydraulic characteristics. Flows can be measured using a staff gauge or a water level recorder calibrated to the weir or flume used.

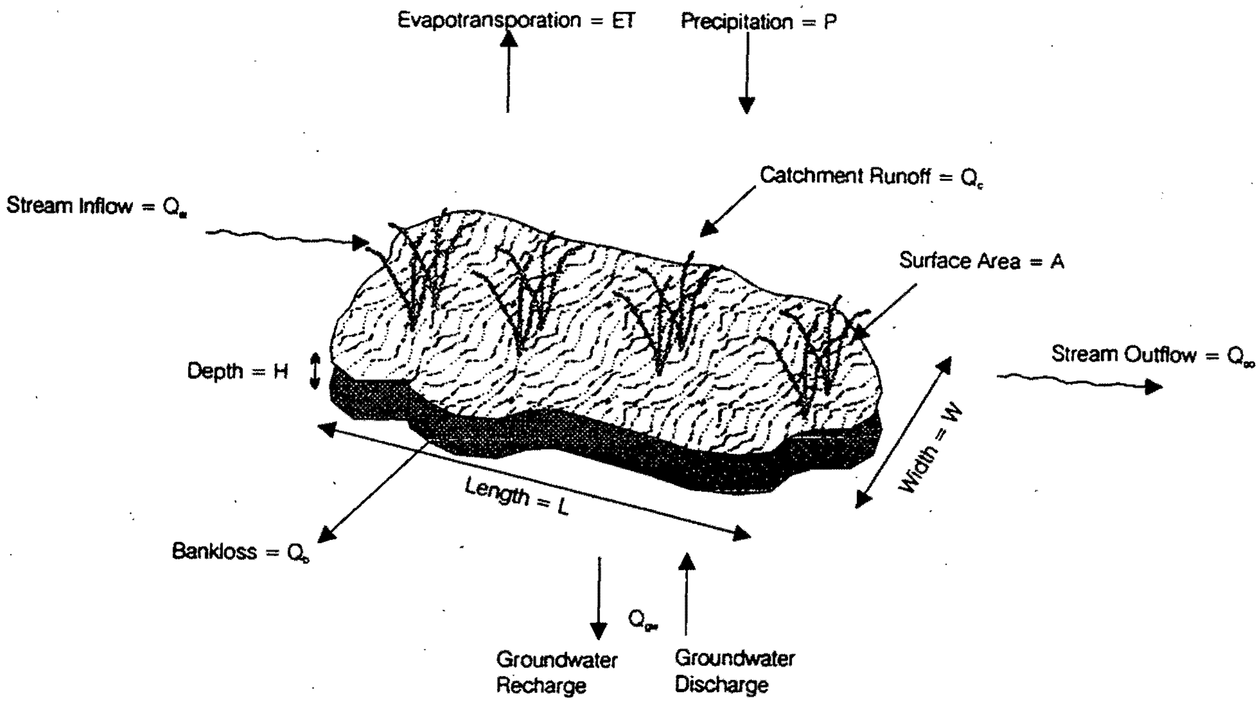



Figure 3. Components of the Water Budget and Associated Terminology

In the absence of a hydraulic control structure, flow measurements can be obtained by relating water level measurements in an open channel to the stage-discharge rating curve developed for that channel. A series of streamflow velocity measurements are made across the channel to obtain the total discharge for a particular water level. Detailed stream flow velocities are measured at stream cross-sections to determine flow volumes. Cross-sections should be marked by stakes put in the ground on both sides of the watercourse. Ten to twenty stream flow velocity measurements are taken across the stream channel, depending upon channel width. A rope can be strung securely across the channel with tape markers at 0.5 m intervals to enable consistent measurements.

Stream flow velocities are measured with a Portable Flowmeter equipped with an electromagnetic velocity sensor. Velocity is measured at depths below the surface equal to 0.2 and 0.8 of the total depth at each station. If the total depth at a station is less than 0.5 m, only one measurement is taken, at 0.6 of the total depth. Cross-sectional and vertical stream flow velocity measurement points are graphically depicted in Figure 4.

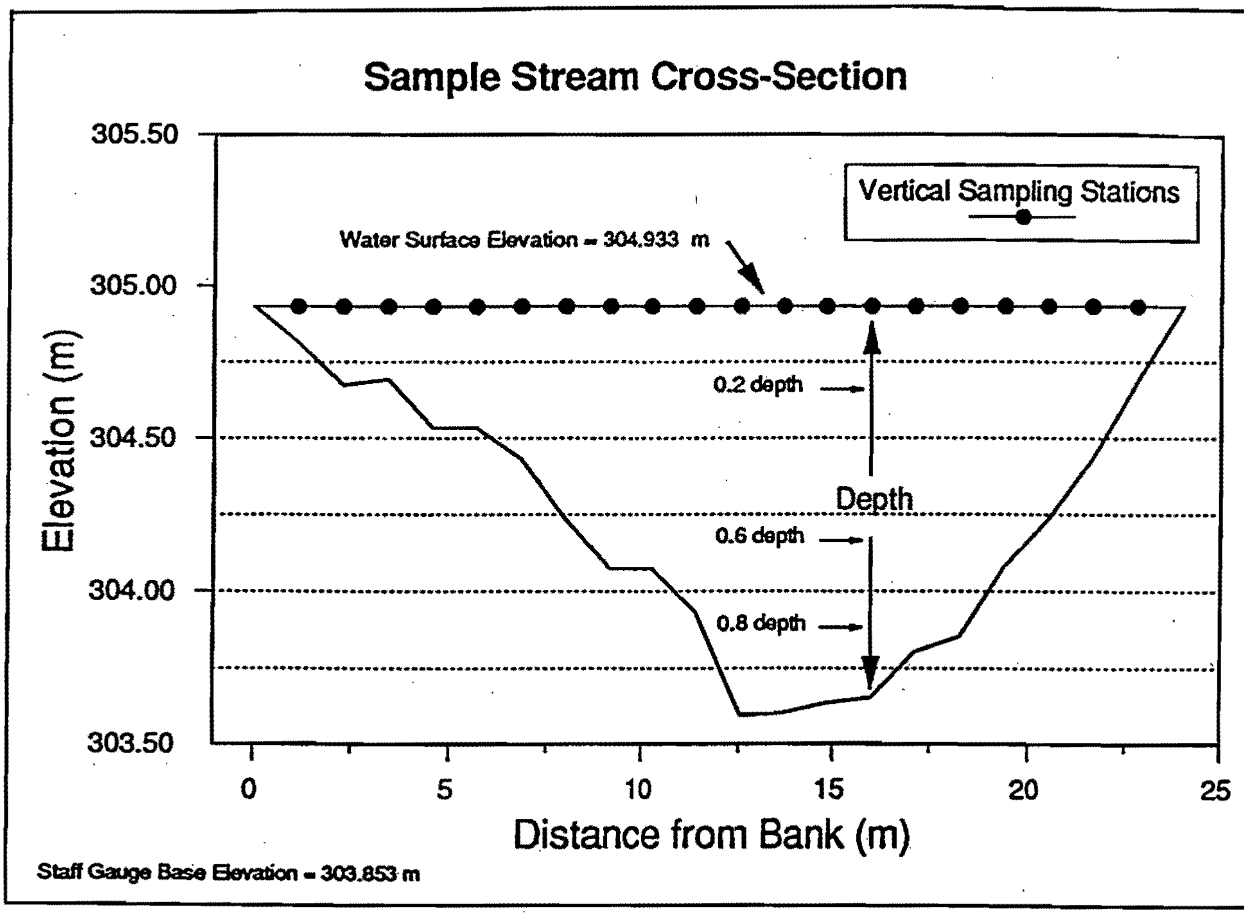



Figure 4. Sample Stream Profile Showing Location of Vertical Sampling Stations and Sampling Depths

To calculate stream flow, the velocity data is entered in an electronic spreadsheet, using a separate file for each sampling station and date. The average velocity (m/s) at each vertical sampling station is multiplied by the area ($m^2$) at that sampling station to obtain the stream discharge rate ($m^3/s$). Areas are determined by multiplying the depth of each vertical sampling station by its representative stream width. The total stream discharge or flow rate is obtained by taking the sum of the products of area times velocity for each vertical sampling station.

A stage-discharge rating curve is prepared by plotting discharge versus water level for a range of water levels. Flow rate at any depth can be determined by measuring the water level, using a staff gauge or water level recorder, and correlating the level to the rating curve. Alternatively, a regression equation describing the best-fit curve through the rating curve can be used to calculate discharge for any water level.

Additional information on the hydrologic characteristics of wetlands, and how to measure them can be found in Kadlec and Knight (1996) and Corbitt (1990).

### 4.2 Landuse/Watershed Basin Characteristics

If a wetland is being used to treat non-point source runoff, a description of landuse within the watershed is useful to guide the chemical parameters to be measured.

For example, in a wetland receiving runoff from an agricultural area, nutrients from fertilizers, bacteria from livestock and pesticides may be the logical focus for monitoring. If the wetland is being used to treat effluent from an industrial facility, the waste stream will likely be characterized which will guide the recommended analysis for sampling in the wetland. If little is known about the inflowing water chemistry, a broad scan of chemicals can be undertaken. However, broad scans for many chemicals can be very expensive, and some effort should be put into trying to focus the monitoring program. This effort will be rewarded by a cost-effective program producing meaningful data.

### 4.3 Inflow/Outflow Water Quality

**Chemical and biological parameters that should be monitored are:**

**A.** Water chemistry and nutrients should be monitored in all wetland types and applications:

- basic nutrients (Phosphorus, Nitrogen, Potassium, Carbon, suspended solids)
- pH, conductivity, hardness
- total suspended solids
- coliform bacteria; chlorophyll a

**B.** Toxic chemicals suspected to occur in effluents depending on the sources within the watershed **(this must be determined on a case by case basis)**, typically these include:

- metals with the highest probability of occuring and which are considered the most toxic (Cu, Zn, Pb, Cd, Ni, Hg)
- chlorinated hydrocarbons such as polychlorinated dioxins/ furans/ biphenyls
- pesticide/herbicide scan or those with highest probability of occurring, depending on adjacent land use
- analysis for any other contaminants suspected to be present due to adjacent land use or discharges into drainage basin (grease; oils; PAHs)

This monitoring should be undertaken within the first year of wetland construction for new wetlands, and as soon as possible for existing wetlands. It is recommended that basic water chemistry is monitored 3-4 times per year preferably after or during storm events and sediment sampling be conducted at least once per year.

### 4.4 Chemical Levels (Exposure Assessment)

#### 4.4.1 Water

**Where and When to Collect**

It is recommended that water samples be collected at the inflow stream, within the wetland itself and at the outlet. By measuring the concentration of inflowing and outflowing water quality, an indication of removal efficiency and loading of certain parameters to the wetland can be obtained.

It is important to collect more samples at the beginning of a monitoring program to establish the parameters of concern and variability of the measurements. Variability can be due to seasonal fluctuations or sampling error (i.e. within station variability). As a rule, duplicate samples should be collected at a minimum of 10-20% of the sample stations. Ideally all water samples are collected in duplicate, but analytical costs may become a deterrent to monitoring at this level of intensity.

At a minimum, water samples should be collected at the inflow, outflow and at least one station within the wetland itself. Samples should be collected on at least four occasions during the year (spring, summer, fall, winter). It would also be informative to collect water samples immediately after a heavy rainfall to determine how chemical concentrations, and hence, loading rates, are affected by storm events. This is particularly important in wetlands treating non-point source runoff, or have large catchment basins. Wetlands directly receiving municipal or industrial effluent are less likely to be affected by rainfall.

#### 4.4.2 Estimates of Loading and Removal Efficiency

When chemical levels in the inflowing and outflowing streams have been established using the methods described in the preceding section, estimates of pollutant loading and chemical removal efficiency can be obtained.

Various terminology is applied to the parameters discussed in this section by different texts and handbooks. Pollutant loading is referred to as flux or mass loading and is a function of the volume of incoming wastewater ($Q_i$ in litres/s or $m^3$/day) and the concentration of the chemical of interest ($C_i$, usually expressed as mg/L). The $C_i$ may be well known if the waste stream is characterized accurately. The inflowing load or mass (Mi) is calculated as follows:

$M_i = Q_i \, . \, C_i$ (Figure 5)

Units may be expressed as g/d; kg/d; kg/yr. The outflow mass will similarly be calculated by:

$M_o = Q_o \, . \, C_o$

The mass removal rate or loading reduction (LR) is calculated as the difference between inflow and outflow:

LR = Mi - Mo (g/d; kg/d)

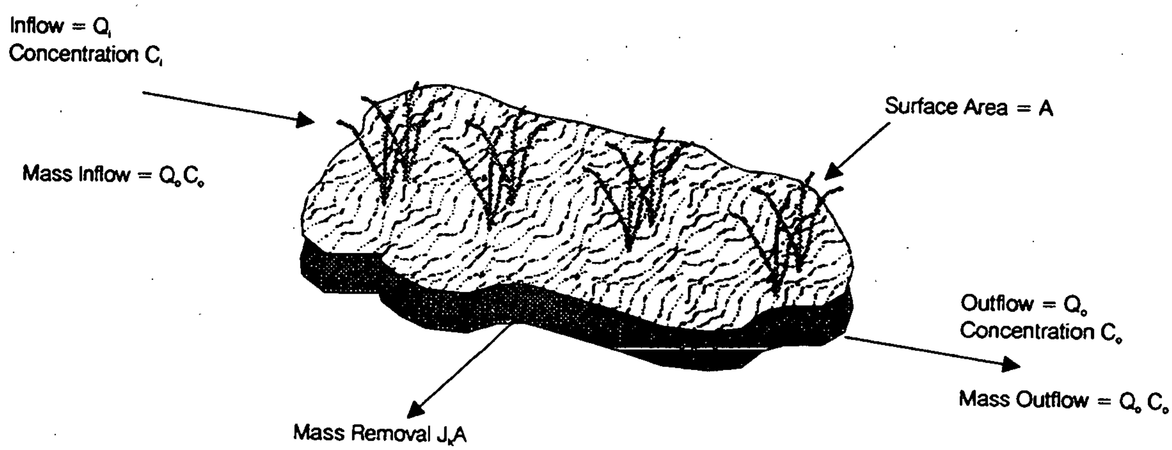



Figure 5. Components of the Chemical Budget and Associated Terminology

### 4.4.3 Sediments

A full detailed chemical analysis of the sediments should be conducted. The surface layer (0-5 cm) will consist of the most recently deposited sediment material and will contain the material most available to biota. Sediment cores can be collected to determine patterns of sediment (chemical) deposition if this would provide meaningful data. Sediment cores can be useful to help establish background concentrations by sampling below the current layer of contamination.

A variety of sampling devices are available which are designed to collect sediments from the bottom of lakes. Either an Ekman Grab or Ponar Grab sampler would be appropriate for surficial sediments and a core sampler is necessary for sampling below the top layers of sediment.

**Where and When to Collect**

The composition of the sediments will largely be governed by patterns of water flow. Fine grained sediments will tend to settle out in quiescent areas, whereas areas of more active flow will generally contain coarser material. Since the spatial variability of sediment samples is generally greater than that of water, samples from different locations in the wetland should be collected.

As a minimum, it is recommended that three stations be sampled, including two sites in the deepest basin of the wetland. If a prominent basin does not exist, choose two stations near the middle of the wetland, or in a quiet portion of the wetland where sediment deposition is likely to occur. One station near the outflow should also be selected. A site near the inflow is less important since sediment deposition is less likely to occur there due to flushing during high water periods.

Unlike water samples, the composition of sediment samples does not vary substantially over time. Therefore, sediment sampling once in the year is adequate to define sediment conditions.

**How many Samples?**
The number of samples to be collected depends upon the size of the wetland, the objectives, the type and distribution of the contaminants being measured, the sediment characteristics and homogeneity, the expected ranges of concentrations of contaminants, the required sample volume and the desired level of statistical resolution. The major practical constraints are the cost of chemical analysis and the logistics of sample collection.

Because sediments tend to be relatively heterogenous compared with water samples, it is recommended that duplicate or triplicate samples be collected at each station for analysis. This will provide a measure of within-station variability. Each sample will consist of at least three samples collected separately and pooled into one sample for analysis.

The volume of sediment required for the chemical analyses depends primarily on the detection limits attainable by the analytical technique. Generally, two litres of whole sediment is sufficient to satisfy the sample size requirements for chemical characterization and contaminant analyses (metals, petroleum hydrocarbons). Before commencing the sampling program the sample end-uses and associated sediment requirements for each analysis should be reviewed with the analytical laboratory and the required volume or weight of sediment calculated.

### 4.4.4 Quality Assurance and Quality Control (QA/QC)

Increasing emphasis is being placed on QA/QC measures both in sample collection and laboratory analysis procedures. The collection of representative, uncontaminated samples in the field is a prerequisite for accurate results. The comments provided here apply to any sample type (e.g. sediments, tissues) being submitted for chemical analysis.

The two most common terms applied to QA/QC are **precision** and **accuracy**. Precision is a measure of the similarity of multiple analyses on a single sample and refers to the reproducibility of the method. Precision can be expressed by standard deviation. Duplicate samples will provide a measure of the precision of the overall program and includes potential error in sampling and sample preparation. It is recommended that at least one in ten samples (10%) be submitted in duplicate for analysis. This will provide an indication of sampling variability and precision of the sampling methods.

Accuracy refers to the agreement between the amount of a chemical measured in a sample and the amount actually present. The accuracy of a method is determined by measuring samples with known quantities present, usually by analyzing standard reference material. Accuracy is generally determined by the analytical laboratory.

There is a vast amount of literature available on QA/QC programs for sample collection and analysis. The best place to start is to speak with qualified staff at the analytical laboratory that you have selected for your program.

In most cases, the contract laboratory will provide the proper sample containers complete with preservatives and necessary instructions for handling and shipping.

### 4.4.5 Toxicity Testing

There is an enormous literature base on techniques for static and flow-through laboratory tests for assessing the toxicity of water and sediments, hence, this topic will not be addressed here (see reviews APHA 1989; Fu *et al.* 1994; Geisy and Hoke 1989). There is also literature available on toxicity testing specific to stormwater runoff (Heaney and Huber 1984; Hall and Anderson 1988; Dutka *et al.* 1991; Dutka *et al.* 1994a, 1994b). Since sensitivity of chemicals varies among species, it is prudent to select several standard tests that assess toxicity in at least one microbial group, and at least one species of invertebrate and vertebrate, usually fish. As a general recommendation, such toxicity tests should be conducted every five to ten years unless contamination in the wetland, based on water and sediment sampling, suggests more thorough and frequent investigation is warranted.

### 4.4.6 When Should Level I Biomonitoring Be Initiated?

As the wetland ages, it will probably become more attractive to wildlife. Nutrient inputs will enhance the development of an invertebrate community, usually followed by an increase in algal/macrophyte community and eventually a vertebrate fauna may use the area. However, over time, the wetland may become more contaminated as organic and/or inorganic chemicals accumulate in the sediments (see Part I, this report). Therefore, baseline monitoring should continue for the 'life' of the wetland/treatment facility. If Provinicial water or sediment guidelines are exceeded; if laboratory bioassays indicate toxicity to biota at any time, then Level II biomonitoring is necessary. Even if indications of toxicity in sediment and water concentrations do not occur, it would be useful to assess the changes in the wetland ecosystem by documenting vegetation and wildlife use over time.

# 5. LEVEL I MONITORING: DEFINING PATHWAYS OF EXPOSURE IN WILDLIFE

Data collected in baseline monitoring will reveal if the wetland is becoming contaminated. Level I monitoring identifies wildlife that may be exposed to contamination, and if warranted, preliminary analysis of biota will be conducted. To summarize, Level I monitoring involves:

- continuation of Baseline Monitoring
- vegetation/ wildlife habitat evaluation
- receptor identification i.e. identify benthic community; wildlife use
- if warranted, contaminant analysis of benthos; fish

## 5.1 Habitat Potential and Populations (Receptor Identification)

### 5.1.1 Wildlife Habitat Evaluation

Most constructed wetlands and stormwater ponds utilize vegetation to help 'absorb' nutrients from the effluent. Generally, habitats that are high in nutrients and/ or encourage wetland and upland vegetation will also attract wildlife. To determine the degree of wildlife use the following habitat parameters should be mapped and evaluated prior to establishing a protocol for monitoring wildlife. However, the monitoring program must be flexible enough so that it can be modified to include unexpected wildlife species. Much of the necessary data to define habitat can be taken from the design drawings for the facility.

1. Determine the area of the wetland that will be less than 30 cm deep. This is the area most likely to support emergent plant species (e.g. cattails (*Typha latifolia*), arrowhead (*Arum* sp.)) and also the area most likely to be used for feeding by dabbling ducks. Map this by season if there are likely to be significant water-level fluctuations.

2. Determine the area that will be less than 60 cm deep. This zone may support robust emergents (e.g. cattails (*Typha glauca*), smartweed (*Polygonum* sp.)), although some may extend into water that is almost a metre in depth.

3. Determine slopes of the banks of the wetland and also of the area under water. Flat slopes will make the area attractive to ducks and waterfowl. Areas with slopes of 2:1 or greater are much less likely to be used by waterfowl. The steeper the slope under water, the less vegetation the wetland will support and therefore wildlife use will be reduced.

4. Determine the vegetation types of the slopes leading down to the water. If tall grass is dominant it may attract ducks, if short mown grass is dominant, it may attract geese and gulls, if tall shrubs or trees are dominant, it will be fairly unattractive to waterfowl and gulls.

5. Examine the vegetation of the adjacent uplands. Manicured areas will attract gulls and geese, tall grassy areas may provide duck nesting habitat.

6. The size of the wetland is important. If it is less than a hectare in area, it will support very limited numbers and species of wildlife. These may include frogs, turtles, probably a maximum of one pair of nesting waterfowl, and possibly a very small population of muskrats. The larger the wetland, the greater the diversity of wildlife that it will support, and numbers of individual species will increase.

The two groups of birds most likely to be attracted to ponds are waterfowl and gulls. Waterfowl includes geese and ducks, and ducks can be further divided into dabblers and divers. Both groups of ducks may also utilize constructed wetlands and ponds during periods of migration. The giant Canada Goose (*Branta canadensis*) nests in southern Ontario and is tolerant of human activity. They frequently nest in urban settings including stormwater management ponds and are essentially non-migratory. They prefer to graze on short grass so golf courses and manicured lawns are favourite feeding areas. Although attracted to stormwater ponds, they feed on terrestrial vegetation and should not be exposed to significant chemicals contained in urban runoff that may accumulate in sediments.

'Dabbling ducks' are those that feed in shallow water. They prefer shallow, calm water generally less than 60 cm deep. Diving ducks on the other hand prefer deeper water and commonly obtain food at depths greater than 3m.

Diving ducks nest on larger bodies of water so habitat in stormwater ponds is generally unsuitable for this group as nesting areas. Most dabbling ducks, with the exception of Mallard ducks (*Anas platyrhynchos*), are fairly intolerant of human disturbance so they are unlikely to nest in urban stormwater ponds. The Mallard is the one duck species that might nest on stormwater ponds but individual ponds would likely only support a maximum of one pair. Preferred ponds would have a vegetated terrestrial 'fringe', be less than 40 cm deep and have an abundance of aquatic invertebrates and vegetation.

### 5.1.2 Aquatic and Wetland Vegetation

The monitoring of vegetation within constructed wetlands is important for gaining an understanding of the potential function of the constructed wetland ecosystem as wildlife habitat. In addition, monitoring vegetation can provide indications of changes to wetland chemistry and hydrology.

The collection and interpretation of quantitative vegetation data is difficult due to the high amount of inherent variability in plant species richness, biomass, productivity and chemical content of vegetation. Therefore, a general survey of wetland vegetation types, associations, forms and a visual survey of vegetation condition is recommended as input to the assessment of constructed wetlands as potential fish and wildlife habitat.

**Step 1: Preliminary Site Assessment**
Aerial photographs (if available) of the constructed wetland and adjacent area should be obtained. Delineate the boundary of the constructed wetland on the air photo. If discernible, delineate inflow and outflow points and the extent of vegetation communities within the wetland, with particular emphasis on the extent of aquatic vegetation communities that may be difficult to survey in the field. Record information regarding land ownership of the constructed wetland and land ownership and land use in the surrounding area.

The above information can be transferred to either an Ontario base map (1:10,000) for large wetlands (i.e. > 20 ha) or a created base map of appropriate scale for smaller wetlands. Determine the size of the constructed wetland.

**Step 2: Field Survey**
<u>Objectives</u>
- confirm location of inflows and outflows
- observe and record drainage within the wetland and immediate area
- delineate wetland boundary
- delineate boundaries between wetland vegetation communities
- determine the vegetation forms within each vegetation association
- record general observations regarding vegetation health

It is recommended that at least two people conduct field surveys in potentially hazardous areas such as large field sites, isolated locations, or areas of muck soils.

**Timing of Field Visits**
For sites with permanent water, field surveys conducted in the summer or early fall are best for obtaining data on the character and extent of submergent and floating vegetation. For wetlands without permanent water, field visits should be conducted in the late summer or early fall to best characterize the vegetation types present. Also, the lower water levels that occur later in the field season provide the most distinct assessment of inflows and outflows to the wetland. The visual assessment of vegetation health should take place during the summer months to avoid observations that may be indicative of plants completing their life cycle in the late summer or early fall.

**Delineation of the Wetland Boundary**
Most constructed wetlands will have well defined boundaries. If not, the wetland boundary can be determined from air photos and transferred to the base map and revised during the field survey. The wetland boundary is drawn where 50% of the plant community consists of upland species.

**Delineation of Wetland Type and Community Boundaries**
Boundaries between vegetation communities within the wetland are determined and mapped during the field visit. The approximate minimum size of a vegetation community to be considered for mapping purposes is based on the size of the wetland:

| Wetland Size | Minimum Community Size |
|---|---|
| up to 1 ha | 0.01 ha |
| 1 - 10 ha | 0.1 ha |
| > 10 ha | 0.5 ha |

Wetland communities are defined by a 4-tier hierarchy (Kavanagh and McKay-Kuja 1992):

- community system: lacustrine or palustrine
- community class: marsh, bog, swamp or fen
- community type: based on major physiognomic aspect of the community
- community association: defined either on the basis of dominant or co-dominant species or habitat descriptors

The boundary of the wetland communities are delineated on the base map through a combination of air photo interpretation and field observation of community boundaries.

Each vegetation community is numbered and a list of vegetation forms that occur in each community created. Each vegetation community may contain one or several combinations of vegetation forms. Following the methods used in the Southern Ontario Wetland Evaluation System (OMNR 1993), any one vegetation form must be present in approximately 25% of a vegetation community to be listed. Dead trees however, should be listed if they cover 10% or more of the community. The dominant vegetation form in each vegetation community should be indicated clearly on the data record.

Special circumstances may require the identification of individual plant species contained within each vegetation community. Special circumstances may include the presence or suspected presence of a wildlife species that utilizes a particular plant species for food, nesting or shelter.

**An example of the information collected from a wetland community definition exercise is as follows:**

**Hypothetical Constructed Wetland**
**System: Palustrine**

<u>Community #1</u>

| | |
|---|---|
| Class | Marsh |
| Type | deep emergent |
| Association | cattail marsh |
| Forms: | re robust emergents (dominant) |
| | ne narrow-leafed emergents |
| | ff free-floating plants |
| | be broad-leafed emergents |

<u>Community #2</u>

| | |
|---|---|
| Class | Marsh |
| Type | shallow emergent |
| Association | shrub-rich marsh |
| Forms: | ls low shrubs (dominant) |
| | ne narrow-leafed emergents |
| | gc herbs (ground cover) |

<u>Community #3</u>

| | |
|---|---|
| Class | Swamp |
| Type | thicket swamp |
| Association | willow swamp |
| Forms: | ts tall shrubs (dominant) |
| | ls low shrubs |
| | re robust emergents |

**Assessment of Vegetation Health**

Visual observations of general vegetation health within each identified wetland community should be conducted during the summer months (July or August). Examine all vegetation forms in each community for widespread or apparently significant signs of stress (i.e. stunted growth, discolouration of the foliage, summer die-back). Should potential stress symptoms be apparent, take several photographs of a whole plant and the affected area up close. If possible, several whole-plant samples (roots, stems and leaves) should be collected, stored in paper bags and refrigerated until they can be submitted to a laboratory for further analysis if required. Should whole-plant collection not be possible (i.e. tall shrubs), collect several branches displaying the symptoms of stress. Substrate samples and water samples should be collected simultaneously adjacent to the vegetation collection. If possible, vegetation, substrate and water samples from an

**Frequency of Sampling**
Vegetation community mapping within the wetland should be first conducted when vegetation becomes well established. Visual assessments of vegetation health should be conducted annually. Wetland community mapping and inventory of vegetation forms present should be repeated every 2 to 5 years.

**Step 3: Data Interpretation**
The vegetation information can be used to focus monitoring protocols for wildlife within the constructed wetland. For example, the vegetation communities and forms present will provide indications of the wildlife species or guilds (e.g. waterfowl) likely to use wetland. Also, the mapping of vegetation communities within the wetland will allow the focusing of wildlife assessment efforts to certain locations containing potential wildlife habitat.

The number of vegetation communities and vegetation forms present in the constructed wetland provide an estimate of the diversity of habitat. Diversity in vegetation communities and diversity in the vegetation forms present provides a variety of habitat types for wildlife.

The wetland community boundaries and vegetation forms in each community should be examined over time for significant community boundary alterations or increases or decreases in types and numbers of vegetation forms present. Changes should be viewed considering natural evolution of the plant communities over time as well as documented changes in water quality or quantity.

Impairment in vegetation health should first be examined with respect to the contaminant(s) the constructed wetland was designed to control. A risk assessment approach should be employed when submitting vegetation samples for laboratory diagnosis of injury or growth impairment to direct the analysis to known potential causes. Vegetation analysis should be conducted in concert with substrate and water analysis.

### 5.1.3 Biota

**Benthic Invertebrates**
Benthic invertebrate communities are often used as indicators of changes in environmental quality (e.g. water or sediment quality, water quantity). For example, severe organic loading usually results in a change in the variety of macroinvertebrates with only the most tolerant ones remaining. Due to the resulting lack of competition, the density of these organisms increases and community diversity is relatively low.

Benthic organisms will bioaccumulate metals and many organic compounds. If there is a dense population they should be collected for contaminant analysis. Bioaccumulation in benthos is the initial step in the contamination of the food web and therefore these organisms are the first indication that there may be hazards to other wildlife.

**Assessing Density and Diversity of Benthos**
Assessing changes in a community over time requires consistent sampling techniques (type of sampler, sieve sizes, preservation and sorting methods) among sampling events. For example, due to life cycle dynamics, there can be a wide variability in community composition and density within a given year. Diversity tends to be lower in the summer months as adults have hatched and larvae are either too small for sieves (are not retained) or are difficult to identify. Therefore, the design of a benthic invertebrate monitoring program must consider the time of year and remain consistent between sample years. Spring or late fall sampling is preferred due to high biomass and maturity of organisms.

At a minimum, three sampling stations should be established as follows: 1) at a central location in the wetland; 2) in the inflow; and 3) below the outflow. High loading of organic material and nutrients in the wetland itself may prohibit the development of a diverse benthic community. Due to the inherent differences in habitat characteristics between flowing water and non-flowing water, invertebrate communities in the wetland should not be directly compared to those in the inflow and outflow of the wetland. Physicochemical characteristics (e.g. water quality, bottom substrate composition, sediment quality, water velocity, etc.) should be similar at stations located upstream and downstream of the wetland such that communities at these stations may be compared. It may also be appropriate to collect samples at various distances downstream of the wetland to determine if water quality is impaired for any distance. The types of organisms present, diversity and number of taxa can be compared to natural streams which would act as a control.

**Collection of Samples for Assessing Diversity and Density**
Samples should be collected in triplicate at each station using the sampler that is most suitable to the site-specific conditions (water depth, velocity, bottom substrate). Surber samplers are the most efficient sampler for flowing streams while a grab sampler such as a Ponar or Ekman would likely be used within the wetland due to water depth and absence of flowing water. Following collection, samples can be sieved and preserved in the field or in the laboratory. Station numbers should be recorded on sample containers with a permanent marker. Any pertinent site information should be recorded on a data form for each station (presence of aquatic vegetation, substrate observations, odour, etc.).

To determine sampling variability, each replicate sample must be preserved, sorted and identified as a unique sample. Mean density and number of taxa can be calculated per station and compared to subsequent surveys. Other diversity indices can also be calculated (e.g. Shannon-Weiner). There are numerous excellent texts and reference books that describe the value of the benthic community and application of monitoring benthos for evaluation of aquatic habitat quality (e.g. APHA 1989; Rosenberg and Resh 1993).

Sorting and identification should be completed by an experienced taxonomist. Data entry into a spreadsheet format facilitates calculation of statistics and data presentation.

**Collection of Samples for Chemical Analysis**

The techniques for collecting samples for analysis are the same as those for assessing diversity and density (above). However, the animals should be not be 'preserved', they should be placed in chemically clean jars (protocol for cleaning jars changes depending on chemical of interest), and placed on ice, preferably dry ice, immediately after collection then stored frozen until analysis. Storage is generally at -20° C or lower temperatures, depending on the chemical of interest.

**Fish**

Fish are useful monitors of the bioavailability of chemicals because, a) they readily accumulate many chemicals from food and water, b) they are often plentiful, c) they are large enough to provide adequate tissue mass for chemical analysis, and d) they are important as food to many wildlife species as well as humans. Therefore, chemical levels in fish provide an indication of potential exposure to other consumer organisms. Relative to other vertebrates, there is a more complete toxicological literature on fish. If there is a low density of invertebrates in a wetland then acquiring enough biomass for analysis may be impractical. An alternative biotic sample can be fish. In some cases fish may be the primary biotic sample of interest.

### 5.1.4 Vertebrate, Non-fish Wildlife

Constructed wetlands have the potential to support a variety of non-fish vertebrate species, depending on the size and configuration of the wetland. However, only a limited number of wildlife species are likely to be permanent residents in constructed wetlands in Ontario. Wildlife inventories should be conducted during the spring, summer and fall during Level I Monitoring. This will produce a list of species that are actively utilizing the constructed wetland. In addition, the vegetation community mapping should be used to identify potential wildlife species that may be attracted to the constructed wetland habitat. Below, selected groups of non-fish wildlife are briefly discussed along with some comment on potential exposure and how they may bioaccumulate contaminants in wetlands.

**Amphibians**

Amphibian species most likely to occur in constructed wetlands in Ontario are American toad (*Bufo americanus*), northern leopard frog (*Rana pipiens*) and green frog (*Rana clamitans)*. In addition, bullfrogs (*Rana catesbeiana*) may occur in wetlands that are a hectare in area or larger, and mink frog (*Rana septentrionalis*) may occur in more northern areas.

Amphibians lay their eggs in the water, the tadpoles develop in the water, and adults respire through their skin. Eggs and tadpoles are most sensitive to environmental contaminants. Green frog and bullfrog are the most aquatic of the frogs and their tadpoles take two growing seasons to develop, so they are exposed to contaminants for longer periods of time. The bullfrog is the most aquatic and sedentary species, spending its entire life in the water. Green frogs may wander away from the wetland in which spawning and transformation occurred, and adult toads and leopard frogs spend much of their adult life in upland meadows.

If bullfrogs are present, they would be a useful bioindicator. Leopard and green frogs and toads are less ideal, but may be more abundant and information could be obtained from eggs and tadpoles. Most adult frogs feed primarily on aerial insects, many of which will have emerged from the wetland. The bullfrog is higher in the food chain and, in addition to invertebrates, it will eat other amphibians, fish and small birds.

**Reptiles**

Turtles are the most common type of reptile likely to occur in constructed wetlands, with snapping (*Chelydra serpentina serpentina*) and painted turtles (*Chrysemys picta picta*) being the most probable species. Some snakes, such as eastern garter snake (*Thamnophis sirtalis sirtalis*) and water snake (*Nerodia sipedon*) may occur in constructed wetlands and the nearby upland areas. Snakes are carnivorous and likely to accumulate lipid soluble contaminants.

Snapping turtles are partially carnivorous, highly aquatic and long-lived, therefore they are susceptible to accumulation of contaminants.

Painted turtles are omnivorous. Young turtles are completely carnivorous, but they eat more vegetation as they age. Typical foods include invertebrates, fish, frogs, carrion, and most species of aquatic plants. Painted turtles are less sedentary than snapping turtles and are more likely to colonize new wetlands.

**Birds**

Groups of birds that may be attracted to constructed wetlands are discussed below, along with their susceptibility to contaminant uptake and suitability as indicators. Virtually all the bird species discussed undertake annual migrations, therefore, it is sometimes difficult to attribute tissue levels in adult birds to specific areas. However, chemical levels in the eggs of many colonial waterbirds and other species such as red-winged blackbirds and tree swallows (*Tachycineta bicolor)* do reflect local exposure conditions and can provide a useful monitoring medium.

The pied-billed grebe (*Podilymbus podiceps*) is the only grebe likely to occur in constructed wetlands, but it is a relatively uncommon species. It eats primarily invertebrates, but will also eat small fish. Because it is a poor flier, virtually all food is obtained from the wetland. The pied-billed grebe migrates to the United States in winter, and therefore can be exposed to other sources of contaminants.

Herons, particularly great blue (*Ardea herodias)* and green (*Butorides striatus*), are likely to visit wetlands to feed on amphibians and fish. They are poorer indicators of environmental quality because they forage in a wide variety of areas; great blue herons may travel as far as 70 km to feeding areas.

Canada geese may nest adjacent to a wetland and feed on vegetation in it. They will also feed on upland vegetation including agricultural crops. Unless the wetland is very large, it is likely to support only a single pair of geese. Larger numbers may occur once the broods amalgamate during autumn migration. Geese are not particularly good indicators of wetland contamination because they may obtain food in a variety of locations, actual use of a wetland may be for only a couple of months, and they may be exposed to environmental contaminants away from the wetland. Since they are herbivorous, they are not highly susceptible to bioaccumulating chemicals.

Dabbling ducks may nest adjacent to a constructed wetland and feed in it. Mallards are most likely to occur, but blue-winged teal and some other dabblers may be present. Drakes are predominantly vegetarians and will eat both terrestrial and aquatic vegetation, but hens eat aquatic invertebrates early in the spring and through the egg-laying phase. Most dabblers have large home ranges of about 500 ha during the nesting season and include a series of ponds within their home range. Wetlands are unlikely to support more than one pair of nesting ducks unless they are more than 4 ha in area. However, the wetland may be used by small flocks of drakes after egg laying is complete. Greatest use of wetlands by dabblers is likely to occur during autumn migration.

Virginia rail (*Rallus limicola*) and sora (*Porzana carolina*) may occur in small wetlands with emergent growth. Food is obtained almost exclusively within the wetland and consists primarily of invertebrates. Rails spend about four months on the breeding grounds and then migrate to the United States. They are very secretive, although their presence can be detected though play-back tapes. In addition, nests are difficult to find.

Spotted sandpipers (*Actitis macularia*) commonly occur at stormwater management ponds. They nest in grasses adjacent to the water, and feed on terrestrial and aquatic invertebrates. They also spend four or five months on the breeding grounds, and nests are difficult to locate.

Gulls are not likely to nest at constructed wetlands, but may visit them to feed on fish and invertebrates.

Tree swallows frequently nest near wetlands and forage on aerial insects, and many of which emerge from the wetland. Tree swallows readily use nest boxes, which makes it simple to monitor contaminant levels in eggs. They lay large clutches and have small home ranges so several pairs can be induced to nest at a wetland.

Marsh wrens (*Cistothorus palustris*) may nest in larger wetlands, typically in extensive stands of cattails. They feed primarily on invertebrates taken from emergents, and it is likely that a high proportion of these will have emerged from the wetland. Four to five months are spent on the breeding grounds, with the remainder of the year being spent in southern climates.

Swamp sparrows (*Melospiza georgiana*) may nest in wetlands with cattail stands. They are primarily granivorous, eating a wide variety of plant seeds, most of which are likely to be on the margin or outside of the wetland.

Red-winged blackbirds are common nesters in wetlands. They are insectivorous and granivorous and may obtain part of their diet outside the wetland or on its fringes.

**Mammals**

The muskrat is the only mammal species that is likely to occur on a regular basis in constructed wetlands. They may also spend their entire life within a small area, so they do have potential as a bioindicator. Muskrats feed almost exclusively on aquatic vegetation, particularly cattails and, to a lesser extent, other emergents. Wetland vegetation is known to accumulate various metals. Studies have shown that muskrats may subsequently bioaccumulate some metals in their tissues.

Numerous other mammal species will visit wetlands to feed and drink. Although they may be exposed to contaminants in this manner, there is no practical method of determining contaminant contributions from a single area.

### 5.1.5 Monitoring Relative Abundance

The text below describes some general approaches to surveying wildlife populations including amphibians, reptiles and birds. Presence, absence and relative abundance are obviously important considerations for evaluating potential exposure to chemicals as well as potential effects. More subtle effects can be monitored by measuring parameters such as egg hatching success, growth and survival of young animals, behaviour, physiology, tissue histology, pathology and egg shell thickness to name a few. Several texts are available that provide good introductions to these methods that could be applied to constructed wetlands (NAS 1979; Keenaga 1979; Lamb and Kenaga 1981; Vouk and Sheehan 1983; Hoffman *et al.* 1995).

**Amphibians**

Environment Canada and the Long Point Bird Observatory have developed a protocol for monitoring amphibians in marshes (LPBO 1996). This entails conducting a three-minute survey listening for calls at each station at least half an hour after sunset on three different occasions. The Marsh Monitoring Training Kit gives explicit instructions on how to set up the stations and when and how to do the monitoring.

This standard protocol is a starting point but more frequent observations are recommended. The survey takes only a few minutes, therefore, it is not expensive compared to some other monitoring requirements. It is suggested that surveys be conducted once a week from mid April until the end of June. This can be modified once the monitoring has been initiated to better reflect the species that are present.

Detailed field methods for surveying amphibians are described in Heyer *et al.* (1994). Initially, basic approaches to confirm presence and absence of species include searching the periphery of the wetland to determine breeding locations. This can be done by looking for egg masses. Because each species has a certain time when it breeds, this survey may have to be done three or four times during mid April to mid July, depending on the latitude of the site. The location of egg masses should be mapped and their abundance should be estimated. This will identify important breeding locations within the wetland for each species, and it will be necessary to know this should level 2 monitoring be required.

**Reptiles**
The inventory should focus on observations or trapping for painted and common snapping turtles.

For presence/ absence information, counts of basking painted turtles are useful for determining the minimum population size. Visit the wetland on a sunny warm day and count the number of basking turtles. Late April through June is an optimum time to do this. Not all turtles will be basking at once, the survey should be done at least three times. The highest count obtained should be considered a minimum estimate of painted turtles inhabiting the wetland.

**Birds**
The Marsh Monitoring Program prepared by Environment Canada and the Long Point Bird Observatory also has a protocol for surveying breeding birds (LPBO 1996). This can be used as a basis for bird monitoring at constructed wetlands, but the techniques should be modified somewhat and the survey should be more holistic. For wetlands smaller than 10 ha, a more intensive inventory is recommended. The location of all birds should be mapped based on an early-morning survey of the entire wetland. The same breeding codes can be used as in the Environment Canada/ Long Point Bird Observatory protocol. This should be done weekly within the time frame recommended in the protocol.

The above survey techniques will provide an minimum assessment of birds breeding in and around the wetland. However, they do not address migrating species and those that may use the wetland as a foraging or loafing area.

There may be a concern that fish-eating birds such as gulls, terns, or osprey will become contaminated if the wetland contains fish that have a high contaminant burden. Gulls will also eat invertebrates that are exposed by declining water levels, but these are unlikely to pose a significant problem since they are a small percentage of their diet. Gulls may use constructed wetlands from mid March until early November. Numbers are likely to peak in July and August once young have fledged and the adults no longer return to the nesting colony daily. Numbers are likely to be greatest from mid morning until mid afternoon unless the wetland is a regionally important feeding area for gulls, in which case numbers will be highest shortly after dawn. To monitor gull numbers, visit the wetland once a week during the appropriate time of day during the period of April through September. If gulls and risk of contamination is a serious problem, gulls can be discouraged through habitat management.

Contamination of migrating waterfowl can be a concern. Spring migration is rapid as birds move to their breeding home ranges, so there is less risk of contamination in spring. The autumn migration is leisurely, and individuals may spend considerable time in one area. The autumn migration period for most waterfowl is from mid August until the end of October, although many may persist right until freeze up. Optimum time of day to survey migrating waterfowl is late afternoon just before the evening feeding flight. Bird numbers can then be estimated while they are on the wetland and while they are in flight. Weekly estimates from mid August until the end of October will give a general indication of the numbers of staging waterfowl using the wetland.

## 6. DATA EVALUATION: HAZARD ASSESSMENT

This is a critical step in the process and requires data interpretation. This step involves preliminary risk assessment to identify Receptors and Exposure. There is no value in proceeding to a Level II monitoring program if a) chemical levels are low, or b) there are no potential receptors i.e. plant/animal life, in the site.

Thus, the first step is to determine if toxic contaminants exist in the basic components of the ecosystem (water, sediment) and/or are bioaccumulating in benthos. If significant contaminants are present, the wildlife inventory results must be reviewed because Level II monitoring may be warranted. A final consideration is the relative abundance of the individual wildlife species. If the population is very low, collection of wildlife tissues for contaminant analysis or pathological examination of animals and their tissues in Level II monitoring may seriously reduce the population size. For instance, if there are frogs in a wetland with contaminated sediments, but there are only three or four females laying eggs, there is unlikely to be a significant risk to wildlife farther up in the food chain, and the collection of amphibians for analysis may be detrimental to the population.

To make rational decisions on whether to proceed to Level II it is useful to evaluate the available information within the framework of an Ecological Risk Assessment (ERA). The conceptual linkages of ERA are illustrated in Figure 6, while the data requirements will be summarized in Table 2. The fundamental information on Exposure and Receptors will have been gathered during the Level I monitoring program. Information on chemical hazard or toxicity will, for the most part, be gathered from the literature, while an estimate of Risk is based upon professional judgement and simple risk assessment methods discussed below.

### 6.1 Exposure Assessment

The data gathered in the monitoring phase will provide quantitative estimates of chemical levels in the environment. Levels of uncertainty can be discussed if sufficient information on data variability has been gathered. Data variability includes ranges (minimum, maximum levels) and standard deviation around the mean. Chemical levels also can vary seasonally and spatially within a system. This information will be used to provide an assessment of chemical transport and fate processes. Fate processes will also be considered within the context of how efficient the wetland is operating. For example, does 100% of chemical X that enters the wetland also exit the wetland, or does some percentage remain within the system? If so, where does it go, into the sediments, water, biota or is it degraded by some process? These questions must be considered when discussing fate and behavior of chemicals.

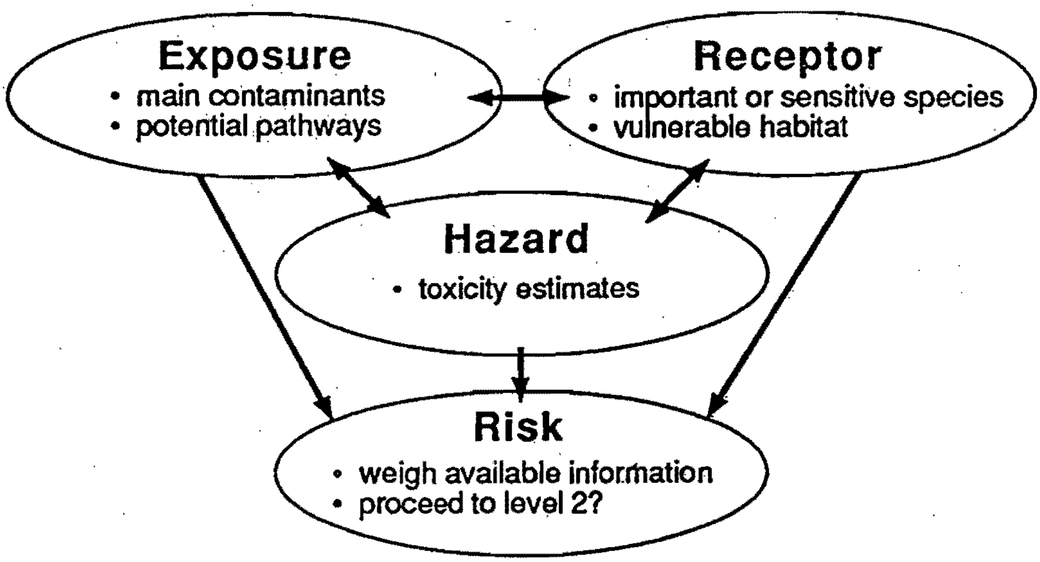



Figure 6. Conceptual Framework for Level I Risk Assessment

A good starting point for evaluating the relative significance of chemicals measured in the wetland is to compare the monitoring results with appropriate criteria such as:

- Provincial water quality objectives (PWQOs) and Fish Tissue criteria (Tables 12 & 13; PART I).
- The Ontario Ministry of Environment and Energy (OMOEE) also has a comprehensive list of Sediment Quality Guidelines that must be consulted (Table 11; PART I).
- If the wetland is classified as an effluent treatment system, site specific effluent limits will be developed under a Certificate of Approval.

As part of the exposure assessment it is useful to develop a conceptual model of the major exposure routes. This is relatively straight forward in a constructed wetland which treats effluent, where the incoming wastewater is the main source. However, it should also take into account the plant and animal species being considered. Is the main route of exposure through sediments, directly from water, or via food chain accumulation?

### 6.2 Receptor Characterization

In this step the most prominent species that are most likely to be affected should be identified (See Level I monitoring). Sometimes these are referred to as Valued Ecosystem Components (VECs). This identification is achieved through the assessment of habitat information on potential wildlife species and from field surveys. Usually at this step the focus is on individual species and consideration of wetland populations. Consideration of communities and ecosystem interactions requires much more detailed analysis than would normally be considered for monitoring constructed wetlands. Life history information should be gathered for the VECs to identify if the wetland provides critical habitat for certain life stages and time periods.

### 6.3 Hazard Assessment

**Canadian Tissue Residue Guidelines (CTRG)**
Environment Canada is currently developing Canadian Tissue Residue Guidelines (CTRG) for wildlife. CTRG will apply to persistent and bioaccumulative chemicals. CTRG are intended to be safe concentrations in the aquatic diets of wildlife to protect, restore and sustain wildlife that consume biota in freshwater, estuarine and marine environments. At present, only the protocol for deriving CTRGs is available (CCME 1996). The protocol is intended as a flexible procedural guide and is not intended to replace best scientific judgement when assessing toxicity. The intent is to provide a benchmark to help interpret biological monitoring data, and assess risk to wildlife, set priorities and interim management targets to measure progress in virtual elimination strategies. These guidelines only apply to a few wetland species such as great-blue heron , green-backed heron, snapping turtle, bullfrog and water snake . Nevertheless the CTRG protocol and the eventual CTRG values will be helpful as a guide to predicting potential toxicity of some chemicals.

**Developing Best Scientific Judgement**
To use the CTRG protocol, and to develop a 'best scientific judgement' in risk assessment it is necessary to gather information from published sources on relative toxicity of the chemicals of concern. For example, if the wastewater entering a constructed wetland contains copper from some industrial process, then toxicity data on copper should be summarized. Furthermore, if the monitoring shows that the copper is being deposited in sediments, information on toxicity to benthos and potential transfer to other organisms should be gathered. If the biological surveys reveal the presence of certain waterfowl that feed on benthic organisms, then data on avian toxicity of copper should also be gathered.

At this stage the focus will normally be on gathering background acute and chronic toxicity data. Toxicity data are usually expressed as the concentration of a chemical either in water, food or sediments that will cause effects among a certain percentage (usually 50%) of the test population. The acute mortality level is referred to as the $LC_{50}$ (lethal concentration) and is expressed as mg/L for water, or mg/kg for sediments or food. For animals, toxicity data are also often expressed as the amount of a chemical (dose) that is required to kill 50% of the animals. This is referred to as the lethal dose or $LD_{50}$ and may be expressed as proportion of food ingested per day per kg of body weight, for example the $LD_{50}$ = 25 mg/kg food/day/kg body weight.

Acutely toxic effects are those that occur within a relatively short time period. For example, a period of 96 hr (4 days) is generally considered acute exposure for fish, while 48 hr is considered acute exposure for aquatic invertebrates. Chronic exposure periods can last for considerably longer periods depending on the species.

Mortality is a relatively easy endpoint to measure, and the consequences for the animal, and subsequently the population are readily understood. More subtle effects can also manifest at lower concentrations in the environment. For example, reproductive success is known to be sensitive to many chemicals. While a certain chemical concentration may not kill an organism, its growth or reproduction may be adversely affected. These more subtle (chronic) effects may require longer (subacute) exposure periods. Therefore, depending on the chemical being reviewed, various toxicity data and values may be available and must be considered. The next section provides an introduction to literature reviews and data on the effects on wildlife of contaminants which commonly occur in constructed wetlands and stormwater ponds.

### 6.3.1 Wildlife Toxicology

Effects of contaminants in wildlife are quantified based on contaminant levels in sediment and water to which wildlife are exposed and/or measuring contaminant accumulation or biochemical indicators of exposure in the animal tissue.

This section is intended to provide a very brief overview of the effects of contaminants in wildlife species that are likely to inhabit constructed wetlands or stormwater ponds. Numerous reviews are listed of the levels and effects of contamination in wildlife. Much of the information has been derived from "Evaluation of wetland biomonitors for the Great Lakes: a review of contaminant levels and effect in five vertebrate classes" by Hebert *et al.* 1993. Such reviews are valuable to the proponent charged with constructing and monitoring the potential ecotoxicological effects of constructed wetland and stormwater ponds. During the past ten years substantial progress has been made in studying the accumulation and effects of toxics to a wide range of wildlife species, but virtually no information was available on contaminant levels or effects for wildlife specifically in stormwater ponds. This impairs our ability to establish the risk to wildlife specifically associated with a 'typical' constructed wetland or stormwater pond. However, the information provided here directs the proponent to methods, expected effects and species potentially exposed and sensitive to contaminants.

The focus is on the levels and effects of organic and persistent contaminants since more data is published on these chemicals. However, less persistent chemicals such as many in-use pesticides, fertilizer by-products, sediments, oils, grease etc. can be toxic to wildlife in wetlands.

When reviewing the literature cited here, readers should be aware that wildlife populations may be more susceptible to environmental contaminants than conventional laboratory animals. The potential toxicity of environmental chemicals is commonly tested on healthy experimental animals having access to unlimited food and water. However, experiments have demonstrated that simultaneous exposure to other chemicals, food restriction, cold or disease can exacerbate the effects of exposure to a toxic chemical (Friend and Trainer 1970; Sanders and Kirkpatrick 1977; Anderson *et al.* 1985; DiGiulio and Scanlon 1985).

### 6.3.1.i Fish

**Occurrence in Wetlands**

Some constructed wetlands are built to provide habitat for fish or to raise fish for human consumption. The feeding behaviour of fish affects the potential uptake of contaminants, and benthic filter feeders such as carp can accumulate high levels of contaminants which makes them unfit for consumption by people or wildlife. Also, carnivorous fish tend to accumulate more contaminants than do herbivorous grazers, therefore, if fish are to be grown in wetlands, the selection of appropriate species may reduce potential hazards.

Florida is probably one of the most advanced jurisdictions regarding stormwater management legislation and implementation of Best Management Practices. Campbell (1994) conducted the only study found which measured contaminant levels in fish collected directly from stormwater ponds. In general, metal levels were higher in fish from stormwater ponds than in control ponds, however, there were differences between species. For example, copper levels were significantly greater in tissues of all three fish species (bluegill sunfish, redear sunfish (*Lepomis microlophus*) and largemouth bass (*Micropterus salmoides*)) collected from stormwater ponds compared with control sites (Table 1). Largemouth bass from stormwater ponds contained significantly higher levels of Cd and Zn, while redear sunfish from stormwater ponds contained significantly higher levels of Cd, Ni, Cu, Pb and Zn (Table 1).

Table 1. **Comparison of mean metal levels in fish collected from stormwater ponds and control sites in Orlando, Florida (data from Campbell 1994)**

| | Stormwater Ponds (mg/kg) | Control Sites (mg/kg) |
|---|---|---|
| **Largemouth Bass (n = 15)** | | |
| Cd | 3.16 | 0.24* |
| Ni | 2.46 | 1.18 |
| Cu | 3.81 | 4.71 |
| Pb | 12.04 | 5.77* |
| Zn | 29.99 | 21.88* |
| **Bluegill sunfish (n = 15)** | | |
| Cd | 0.006 | 0.004 |
| Ni | 0.156 | 0.046 |
| Cu | 2.08 | 1.07* |
| Pb | 0.77 | 0.54 |
| Zn | 36.61 | 30.72 |

* values are significantly different at $p < 0.05$

**Contaminant Effects**

The chronic effects of persistent contaminants on Great Lakes fish physiology have been reviewed (Cairns *et al.* 1984; Fitchko 1986; Government of Canada 1991). The science of aquatic toxicology is well established and the general literature available on this subject is vast. Several excellent reviews and recent textbooks are available (eg. Rand 1995). The most sensitive physiological responses of fish to toxic chemicals are generally survival, growth and reproductive success. Other, but generally more difficult to measure, responses include changes in oxygen consumption, breathing rate, coughing response, metabolism, enzyme activity, thyroid activity, heart rate, histopathology, morphology, immunology, locomotor activity, feeding, temperature selection and other behavioural traits.

Many abiotic and biotic factors are known to modify chemical toxicity to fish including life stage, pH, water hardness, temperature, disease and parasitism, diet, dissolved oxygen, chlorides and exposure to sunlight. The variables are likely to influence the impact of site specific conditions on aquatic biota.

**Suspended Solids**
Suspended solids will comprise one of the major constituents of stormwater ponds in the water column particularly after a storm event. This is not surprising since the primary purpose of constructed wetlands and stormwater ponds is to provide an area of low energy to permit accumulation and settling of suspended solids prior to the runoff water entering the nearest natural surface waterbody. Therefore, suspended solids pose a potential threat to aquatic biota that rely on gills for respiration.

There has been a considerable amount of research conducted on the effects of suspended sediments to fish and aquatic organims. The concentrations of suspended solids can become increased due to a wide range of human activities including mining, forestry, agriculture, road construction and in-stream construction activities such as pipeline crossings.

**The principal methods that inert sediments can affect fish are by:**

1. direct effects on free-living fish either by killing, reducing growth rates or reduced resistance to disease
2. interfering with the development of eggs and larvae
3. modifying natural movements and migration
4. reducing the abundance of food organisms
5. reducing the efficiency of catching prey items

(after Alabaster and Lloyd 1982).

The general literature suggests that fish will naturally avoid elevated concentrations of suspended sediments before significant effects can occur. Avoidance may be a practical reaction in large lakes and rivers, but it is unlikely that stormwater ponds or constructed wetlands would provide adequate protection areas of clean water during a storm event unless these were specifically incorporated into the pond design.

Newcombe and MacDonald (1991) recently reviewed over 70 papers dealing with the effects of suspended sediments (SS) to fish. The prime result of their findings was that duration of exposure to SS must be considered when attempting to predict the impacts of SS to fish. The concentration of SS alone was not a good measure of the potential effects of SS to fish and aquatic biota. Furthermore, toxic effects are a function of both concentration and exposure. Short term exposure to elevated SS should not pose a risk to fish, but only where avoidance to clearer water is possible.

**Summary**
The available toxicological literature regarding fishes is vast. Fish are sensitive to a number of chemicals known to be present in stormwater ponds including pesticides and metals, particularly zinc and copper. Elevated water temperatures and fluctuating water levels may also

limit the types of fish species found in stormwater ponds, but may not pose the same constraint in wetlands.

Fish can bioaccumulate lipophilic contaminants including methylmercury, PCBs and pesticides. Lead, copper and zinc may accumulate in specific tissues such as bone, liver or kidney, but not generally to levels that would pose risk to a consumer organism. From the literature reviewed, mercury has not been reported as a common constituent in urban runoff, therefore, there is low risk of it accumulating in fish inhabiting these environments. Bioaccumulation of organic contaminants is more likely to represent a potential risk to predator species.

#### 6.3.1.ii Amphibians

A database on the toxicity of chemicals to amphibians was recently (1995) developed by Mr. Bruce Pauli, Canadian Wildlife Service, National Wildlife Research Centre, Hull, Quebec, this information is also avalilable on the internet at: http://www.cciw.ca/green-lane/herptox. An earlier review of the literature is available from the Canadian Wildlife Service, Burlington, Ontario (Harfenist *et al.* 1989). Sources such as this should be consulted by the interested reader.

The Frog Embryo Teratogenesis Assay: *Xenopus* (FETAX) is a 96 hour, whole embryo bioassay which utilizes eggs/embryos of the African Clawed Frog (*Xenopus laevis*) exposed to known concentrations of water soluble chemicals. It has been widely used in the laboratory to evaluate the embryotoxicity, teratogenicity and effect on growth of single and complex mixtures of chemicals in solution (Dawson and Schultz 1989; DeYoung *et al.* 1989, 1990; Fort *et al.* 1989, 1991; Dawson *et al.* 1990, Rayburn *et al.* 1991; Linder *et al.* 1990).

#### 6.3.1.iii Reptiles

There is little documented information about the presence of reptiles in constructed wetlands other than the fact that they may colonize wetlands and add to the faunal diversity. There is potential for turtles to use embankments and dikes to lay eggs or hibernate.

Meyers-Schone and Walton (1994) have reviewed the literature on contamination and effects in turtles. However, there is almost no data on the effects of contaminants on snakes. There are a number of recent studies which have determined that high concentrations of organochlorine pesticides, polychlorinated dioxins, furans and biphenyls can be accumulated in tissues and eggs of the common snapping Turtle (Stone *et al.* 1980; Helwig and Hora 1983; Olafsson *et al.* 1983; Ryan *et al.* 1986; Bryan *et al.* 1987; Struger *et al.* 1993). This long-lived, omnivorous species is the largest turtle found in Ontario (Hammer 1969; Conant 1975; Lovisek 1982) and one of the most common (Weller and Oldham 1988). Most animals remain in the same home range in consecutive years (Obbard and Brooks 1981). Estimates of home-range size vary from 3-4 ha (Obbard and Brooks 1981) to 0.8-28.4 ha (Pettit *et al.* 1995). It is consumed by humans (Lovisek 1982), hence, the potential for contaminant transfer to humans has been the focus of several studies (Hebert *et al.* 1993). Common snapping turtle eggs are sensitive to polychlorinated dioxins, furans and biphenyls and exhibit deformities in embyros and hatchling turtles and higher rates of unhatched eggs (Bishop *et al.* 1991).

### 6.3.1.iv Birds

Birds, particularly waterfowl, are attracted to wetlands of all sorts, and there is the potential for them to accumulate contaminants if they are present in wetlands. Contaminants may be obtained from the food they eat or from sediments or water which are also ingested during feeding. Birds breeding on constructed wetlands for industrial and stormwater management have reproduced well in North Dakota and large numbers of ducks and geese have bred on specially created islands in these wetlands (Litchfield and Schatz 1989).

Avian reproductive failure has been attributed to the effects of heavy metal contamination (Scheuhammer 1987; Kraus 1989) and pesticide uptake (Adamus and Brandt 1990). Late season accumulation of dieldrin in sentinel mallards was documented at a sewage lagoon in Winona, Ontario, probably due to wastes from vineyard operations. The levels accumulated were below those previously reported and below levels thought to be critical for raptors (Gebauer and Weseloh 1993).

In a study in which wing-clipped mallards were introduced and permitted to feed in contaminated sites as well as at control ponds the mallards accumulated PCBs, organochloride pesticides and metals. After periods of up to 100 days residence on the waterbodies, all concentrations in the birds were below levels believed to have harmful effects on birds. However, ducks collected in Hamilton Contaminant Disposal Facility (CDF) had concentrations of PCBs 5300 times more than at day "0" and exceeded Health and Welfare guidelines of 0.5 $\mu$g/g lipid weight and USFDA guideline of 3 $\mu$g/g for poultry (USFDA 1979; Gebauer and Weseloh 1993).

There was also a difference between resident and migratory ducks (mallard and redhead (*Aythya americana*)) feeding on contaminated marshes of Walpole Island, Ontario. Liver and muscle concentrations of octachlorostyrene (OCS) were 115 $\mu$g/kg, hexachlorobenzene (HCB) 30 $\mu$g/kg, and pentachlorobenzene (QCB) 1.5 $\mu$g/kg. The concentrations in migratory species were 56, 8.7 and 0.4 $\mu$g/kg respectively (Hebert *et al.* 1990).

Waterfowl production was affected due to chemical contaminants in a major wetland in central Arkansas. Reproductive impairment was reported in wood ducks (*Aix sponsa*) from wetlands contaminated with dioxins and furans (White and Seginak 1994). The wetland in central Arkansas was contaminated by a former chemical plant that manufactured the herbicide, 2,4,5-T. A study revealed higher concentrations of toxic dioxin and furan isomers in eggs of wood ducks closest to the source, compared with reference eggs. Nest success, hatching success and duckling production were all decreased at sites near the source of contamination. The threshold for the start of toxic effects on reduced productivity was 20-50 ppt (TEQs - Toxic Equivalent Quotients) in eggs.

The effects of PCBs, PAHs and dioxins on fish-eating birds were recently reviewed by Bosveld and Van Den Berg (1994). The authors provide succinct summaries of the concentrations of these chemicals measured in bird tissues from numerous studies including the Great Lakes. The effects of these compounds on fish-eating birds include reduced hatching success, embryonic deformities, growth retardation, deformed embryos, eggshell thinning and altered cytochrome P-450 synthesis. The lowest observed effect levels (LOELs) associated with the different toxic and biochemical end points were also summarized.

**Summary**

Birds that may be expected in stormwater ponds and constructed wetlands can be sensitive to the chemicals that may occur in these habitats (e.g. lead, pesticides). In addition to direct toxicity, reduced food availability may affect waterfowl growth and reproduction as an indirect effect of chemical contaminants. Therefore, the issue of length of exposure to certain chemicals becomes important when trying to determine risk. There is no formal documentation on the amount of time spent by different bird species in either stormwater management ponds or constructed wetlands. Of the two, constructed wetlands generally provide habitat more amenable to longer visits and greater use by waterfowl. Stormwater ponds are generally much smaller, are in more urbanized or populated areas and contain less vegetation for cover, all features which discourage extensive use by birds.

### 6.3.1.v Mammals

Metals are a common contaminant in constructed wetlands and stormwater ponds and accumulation of these contaminants is well documented in wild mammals. Effects of metals are well known in mammals however their are fewer documented effects in wild mammals.

### Lead (Pb)

Lead is commonly found in urban runoff. Lead accumulates in the mammalian body so that chronic exposure to small amounts can cause toxicosis. Absorption of ingested Pb is relatively low but a large portion of the absorbed Pb is retained in soft tissues initially and later in the bone (Buck *et al.* 1976). Pb intoxication generally results in anemia and anorexia. Anemia results from impaired heme synthesis and shortened life span of red blood cells (Goyer and Mahaffey 1972). Circulating Pb combines with erythrocytes and destroys red blood cells, and bone marrow is affected so fewer red cells are produced. In the kidney, Pb causes degeneration and necrosis of renal tubule cells.

Raccoons may be found at constructed wetlands or stormwater ponds. The liver Pb levels in 14 wild raccoons collected from various areas in Connecticut was <1.0 - 20.0 µg/g (Diters and Nielson 1978). Microscopic brain lesions were noted in 3 raccoons with liver Pb levels over 10 µg/g. Liver or kidney Pb levels greater than 5-10 µg/g are considered toxicologically significant (Buck *et al.* 1976; Frank *et al.* 1981). Although raccoons are able to tolerate a large body burden of Pb without noticeable effects, stress in the animal such as disease, inadequate diet, or diets low in calcium may lower the required toxic dose (Sanderson and Thomas 1961). This speculation is supported by studies which show Pb lowers resistance of mice to bacterial infection at exposure levels not causing other signs of poisoning (Selye *et al.* 1966; Hemphill *et al.* 1971).

Sanderson and Thomas (1961) measured tissue Pb levels as part of an investigation into declining raccoon populations in Illinois. The mean liver Pb concentrations of 101 raccoons killed by hunters was 6.8 µg/g (range 0-32 µg/g). Results of the study did not clearly relate Pb body burdens to biological effects.

## Arsenic (As)

The biochemical characteristics and toxicity of As compounds vary considerably. In general, the organic forms of As are less toxic than the inorganic forms, and the pentavalent compounds are less hazardous than trivalent (NAS 1979). The organic As compounds phenylarsonic acids and their salts are commonly used as feed additives to control disease and promote growth of livestock (Buck *et al.* 1976). The two principal sources of environmental As contamination is from the use of As in pesticides and herbicides, and emissions from metal smelters.

Field studies by Boyce and Verme (1954) showed that arsenic (sodium arsenite) was lethal to deer when licked from the bark of treated trees. The minimal lethal dose was about 34 mg/kg.

The reported levels of As in wild mammals are generally very low (eg <0.3 µg/g) (Smith and Rongstad 1981; Radvanyi and Shaw 1981).Woolf *et al.* (1982) suggest that acute or subacute As poisoning may be associated with levels as low as 2 µg/g in liver or kidney, however, levels above 10 µg/g are usually accepted to confirm As poisoning (Buck *et al.* 1976).

## Cadmium (Cd)

There are very few reports of Cd-induced injury to wildlife. Incidents of Cd poisoning in humans and domestic animals have frequently been overlooked or incorrectly diagnosed as something else due to the multitude of interactions of Cd with other essential and toxic elements (NRCC 1979). The main clinical signs of Cd toxicity in animals are anemia, retarded gonad development, enlarged joints, scaly skin, liver and kidney damage and reduced growth. These signs are similar to a primary Zn deficiency. Furthermore, Pb is often present with Cd in environmental samples or industrial applications, and biological effects are most often attributed to Pb.

A field study by Herbert and Peterle (1990) showed that raccoons accumulated high organ levels of both cadmium and PCBs without signs of any obvious deleterious effects. They suggest that because of this species apparent resistance to certain contaminants and the fact that it depends on aquatic systems for its food source, raccoons are a potentially important indicator species for environmental contamination. Everett and Anthony (1977) found that Cd levels in muskrat tissues were related to Cd in plants and suggest that the muskrat is also a valid indicator of Cd pollution in aquatic ecosystems. Erickson and Lindzey (1983) reported adult muskrat Cd levels were significantly higher than in juvenile muskrats. Aquatic macrophytes can accumulate cadmium (Miller *et al.* 1983) and may act as a source of cadmium to grazing herbivores such as muskrats.

Cadmium is the only metal which clearly accumulates with increasing age of the organism. Kidneys are the preferred site of Cd accumulation in mammals. Significant accumulation of cadmium has been demonstrated in wildlife exposed to point sources of cadmium (Sileo and Beyer 1985; Gunson *et al.* 1982; Hunter *et al.* 1984a, 1984b; Munshower 1972).

### Copper-Molybdenum (U-Mo)

Copper is commonly reported in urban runoff at elevated concentrations. Problems related to Cu-Mo metabolism are widely reported in grazing domestic livestock, and there are some reports of concern for wildlife (Robbins 1983; Ward and Nagy 1976; Flynn *et al.* 1977). The metabolism of Cu, Mo and inorganic sulfate is extremely complex and interrelated (Underwood 1977). The interactions of Cu-Mo can result in two toxic scenarios; excess Cu-deficient Mo, or deficient Cu-excess Mo. In the presence of inorganic sulphur it is impossible to delineate between the toxicity of one and deficiency of the other (Buck *et al.* 1976). Deficiency or excess of Cu-Mo are most prominent among ruminants and directly related to Cu-Mo balance in soil and forage.

Copper levels may be elevated in aquatic vegetation near point sources, but tissue Cu levels in muskrats are low (Everett and Anthony 1977; Radvanyi and Shaw 1981). Wren *et al.* (1988) reported elevated levels of copper in organs of otter and mink near Sudbury compared with other areas in Ontario.

### Zinc (Zn)

High zinc concentrations have been measured in water and sediments of stormwater management ponds. Elevated tissue Zn levels can be found in wild animals living near local point sources of Zn emissions. Everett and Anthony (1977) reported high levels of Zn in sediments, vegetation and muskrats from a marsh receiving drainage from a Pennsylvania Zn mine. Liver Zn levels in muskrat from contaminated and control sites were 81.2 µg/g and 35.0 µg/g, respectively.

## 6.4 Risk Characterization

The relative risk to biota in constructed wetlands is undertaken by consolidating all the information on exposure, receptors and relative toxicity (hazard) of chemicals and species involved and making an assessment. Because there are no specific and standardized acceptable concentrations of environmental contaminants in wildlife the assessment may be partially qualitative at this stage to predict potential risk to wildlife. This may be based on professional judgement/ wildlife toxicology literature. But, if possible, a simple quantitative assessment should be undertaken.

The simplest quantitative risk characterization is the quotient method. The quotient method uses reliable methods of exposure and toxicity data for either a site specific receptor, or a suitable surrogate species. The quotient method derives its name from the simple mathematical approach used to derive a quotient as follows:

Q = Exposure Concentration/Hazard Level

Using a simple hypothetical example, monitoring in a wetland may produce the following information:

- concentration of chemical X in water = 25 mg/L
- the concentration of chemical X that is acutely toxic (eg. the $LC_{50}$) to rainbow trout = 2.5 mg/L

Although the wetland contains catfish, not rainbow trout, the trout toxicity data are used as the nearest fish surrogate species. Using the quotient method:

$$Q = 25/2.5 = 10$$

Where, Q >1, there is potential risk to the species under consideration, if Q <1, there is no estimated potential risk.

The quotient method is simple and a good first approximation. The estimate of risk is obviously dependant upon the reliability of the data used in the equation, and the reliability of the estimate becomes more judgemental as Q approaches 1.0. The estimate of risk can be expressed as a range of values using ranges of exposure concentrations over the monitoring period, or ranges of toxicity (hazard) levels.

**Table 2 Summary of Data Requirements for Level One Ecological Risk Assessment (modified from Gaudet *et al.* 1994)**

**EXPOSURE ASSESSMENT**

- qualitative, preliminary quantitative methods
- based largely on monitoring data

♦ Section of Target Chemicals
- select target chemicals based on review of water and sediment data

♦ Chemical Transport and Fate
- provide preliminary quantitative estimates if possible

♦ Exposure Pathways Analysis
- identify most important exposure pathways

♦ Aquatic or Terrestrial Exposure
- identify most important exposure pathways/food chains
- provide preliminary estimates of exposure or tissue concentration using BCF, BAF

♦ Uncertainty Analysis
- identify data gaps

♦ Output - preliminary, quantitative estimate of exposure via dominant pathway(s)

**RECEPTOR CHARACTERIZATION**

- qualitative, preliminary methods
- based largely on site visits, and evaluation of habitat potential

♦ Identify Receptors
- identify habitats, species and populations

♦ Select Endpoints
- select assessment and measurement endpoints with focus on individual and population levels
- ensure priority receptors are emphasized

♦ Relate to Exposure Assessment
- assess possible spatial/temporal overlap of receptors and contaminants of concern

♦ Output - basic life history information on species identified as potential receptors

**HAZARD ASSESSMENT**

♦ Hazard Identification
- review existing site data (chemistry and effects)
- review toxicity of Contaminants of Concern identified in exposure assessment

♦ Endpoints
- select measurement and assessment endpoints
- choose species for which toxicity data are readily available (extrapolate to VEC)
- focus on acute endpoints (e.g. mortality); collect chronic/sublethal information simultaneously

♦ Output - LC50, LD50, benchmark concentrations for selected chemical and species

**RISK CHARACTERIZATION**

- qualitative and quotient methods
- characterize risk as "high", "intermediate", "low"
- estimates of uncertainty restricted to safety factors
- identify key uncertainties, data gaps

# 7. LEVEL II MONITORING: QUANTIFYING PATHWAYS AND ASSESSMENT OF EFFECTS

The Level II program contains several options for monitoring. Detailed Level II monitoring will be site specific and tailored to the individual wetland. Study design advice should be obtained from a wildlife expert before beginning a Level II monitoring program. The following sections provide a brief summary of methods that may be employed to quantify identified pathways and assess the effects of contaminants exposure in a Level II investigation.

## 7.1 Exposure Monitoring

There is a substantial body of literature available on biomonitoring techniques that can be applied to constructed wetlands and many of these references are refered to in the Wildlife Toxicology section (above). Supporting references for specific techniques are provided in the relevant sections below. General documents that provide information on methods for measuring contaminant uptake (exposure) and assessing the health of individuals, populations or communities include Environment Canada (1991a, 1991b), Hunn (1988), Newman (1995), Rand (1995) and Hoffman *et al.* (1995) to name a few.

**Vegetation**
Aquatic vegetation is known to accumulate certain chemicals, particularly metals. However, there are substantial differences between species and groups of plants (Miller *et al.* 1983). Furthermore, some species, such as cattails (*Typha* sp.) tend to accumulate metals in root tissue to a greater degree than other parts of the plant (Knowlton *et al.* 1983; Taylor and Crowder 1981). Therefore, consistent sampling of tissue and plant type is important.

As constructed wetlands become used more frequently, information from existing case studies will become more available. A number of studies have specifically examined nutrient and chemical uptake in constructed wetland vegetation to help determine removal efficiencies and environmental fate (Dobberteen and Nickerson 1991; Ansola *et al.* 1995; Rai *et al.* 1995; Peverly *et al.* 1995).

**Fish**
See comments in Section 5.1.3.

**Caged Clams and Fish**
Methods for collecting resident populations of benthic organisms and fish were presented earlier in this report. Animals captured this way can be analyzed for body burdens of contaminants. There are some limitations of using wild animals that can be overcome by using caged specimens that are placed in a waterbody and exposed to ambient conditions. Animals of uniform size, age, species and background tissue levels are held in appropriately sized cages and placed in the constructed wetland. Mesh size should be adequate to allow full water circulation and possibly food passage. Freshwater clams are relatively simple as they are filter feeders and require no additional nourishment. Fish may require supplemental feeding depending on the species and size being used.

The biomonitors must be left *in-situ* for a sufficiently long period to ensure that contaminant accumulation does take place. The rate of uptake will be species and chemical dependent. The experiment could be run during summer months when warmer temperatures will increase metabolism and rate of uptake. An exposure period of 4 to 8 weeks should be adequate to monitor the bioavailability of chemicals in the wetland. Tissue levels and body size should be measured at the beginning of the experiment (T=0) and at the conclusion of the test. More detailed information on growth rates and uptake kinetics can be acquired by sampling throughout the experiment but this additional information is optional. Caged clams and fish have been successfully used to monitor chemical uptake and bioavailability in a number of aquatic ecosystems (Curry 1977; Davies *et al.* 1979; Forester 1980; Flood *et al.* 1986).

**Amphibians and Reptiles**

Amphibians (e.g. frogs) and reptiles (e.g. snapping turtles) are two other groups that have received increasing attention and could be collected for tissue analysis. Frog eggs are considered sensitive to some environmental contaminants. Assesment of effects in wildife species can be performed using field enclosures to contain eggs and/or tadpoles (Harris 1996) and/ or can be exposed to field-collected water/ sediment samples (Harris 1996). Collect a subsample of frog eggs from several different egg masses, and do this for as many species as possible. For snapping turtle eggs, in particular, good comparative data from a number of studies in Canada and the USA are available. The use of turtles as biomonitors of environmental contaminants has received considerable attention in the past decade. As such, there are a number of excellent papers describing their value and methodology in more detail (Olafsson *et al.* 1983; Struger *et al.* 1993; Meyers-Schone and Walton 1994; Albers *et al.* 1986; Bishop *et al.* 1991; Bonin *et al.* 1995).

**Birds**

Novel approaches may be required to obtain a sufficient number of relevant samples of birds exposed exclusively to contamination in a constructed wetland. For example, the use of captive mallards, clipped to prevent movement away from the waterbody is possible (Gebauer and Weseloh 1993; Custer *et al.* 1996). Mallards feed on aquatic invertebrates and may be good indicators of ambient chemical availability. Chemical accumulation could be monitored in eggs, and if warranted, followed through fledglings, juveniles and adults. Numerous studies are available that describe contaminant monitoring in wild birds (Ohlendorf *et al.* 1987; Sundlof *et al.* 1994).

Tree swallows and red-winged blackbirds feed extensively on emerging insects and reflect very localized conditions (Bishop *et al.* 1995). Tree swallows are very common and construction of nesting boxes will often attract tree swallows to nest near the desired waterbodies for study. Invertebrates on which they feed and eggs could be sampled in the spring for chemical analysis (Nichols *et al.* 1995). Comparative data are also available for these species from other locations (Bishop *et al.* 1995). It is likely that nesting boxes will have to be erected and maintained in order to have a nesting population of tree swallows adjacent to the wetland. Thus, a conscious decision must be made that the results of monitoring tree swallows will be significant enough to warrant artificially exposing these birds to contaminants.

## 7.2 Wildlife Health Assessment

A wide variety of techniques have been developed to assess wildlife health by measuring different parameters in populations or individuals. Specific tests to assess effects of individual chemicals are available for only a few substances. Therefore, general assessment of health may be necessary (Table 3; see also Wildlife Toxicology review above). Generic techniques can be used to assess organism response to a range of stresses or chemical substances. Biotic variables such as growth rate or reproductive success (fecundity, egg clutch size, hatchling success), immunosuppression and endocrine function would be examples of non-specific but important responses. Other end points may be quite specific for a particular substance. For example, the enzyme ALAD is sensitive primarily by lead, while cholinesterase activity in birds is affected by carbamate and organophosphate pesticides. Examples of techniques used to assess ecosystem health are provided in Table 3. The list is not exhaustive, as entire texts have been devoted to this subject area. Recent reviews (Adams 1995; Hoffman 1995; Melancon 1995) are available on wildlife ecotoxicology and approaches to assessing effects of contaminants on free-living wildlife. These reviews, and an experienced wildlife toxicologist should be consulted for the appropriate design for each case.

**Table 3 Select Techniques Available to Assess Wildlife Health**

| Group | Variable/Parameter Measured |
|---|---|
| Benthic Invertebrates | Number of organisms per site<br>Number of taxa per site<br>Diversity index<br>Chironomid mouth part deformities<br>Sediment toxicity bioassays using Hyalella, Chironomus or Tubifex |
| Fish | Growth rates<br>Fecundity<br>Liver Somatic Index (LSI)<br>Gonad Somatic Index<br>External lesions/abnormalities<br>Skeletal deformities<br>Metallothionein in liver, kidney (metals, e.g. cadmium)<br>Tissue analysis for specific chemicals<br>Liver Mixed Function Oxidase (MFO) activity<br>ALAD[1] activity (lead)<br>Histopathology<br>cholinesterase inhibition<br>sex steroid concentrations |
| Birds (some tests may apply to amphibians, reptiles or mammals) | Nest clutch size<br>Hatching success<br>Tissue/plasma analysis<br>fledgling success and growth<br>Cholinesterase Inhibition<br>Liver Cytochrome P450 level<br>Eggshell thickness<br>gross deformities<br>ALAD activity<br>Histopathology<br>GSSG/GSH[2] ratio<br>Metallothionein levels<br>Metabolites in bile |

[1] ALAD = Aminolevulenic Acid Dehydratose
[2] GSSG/GSH = Oxidized glutathione/reduced glutathrone ratio

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