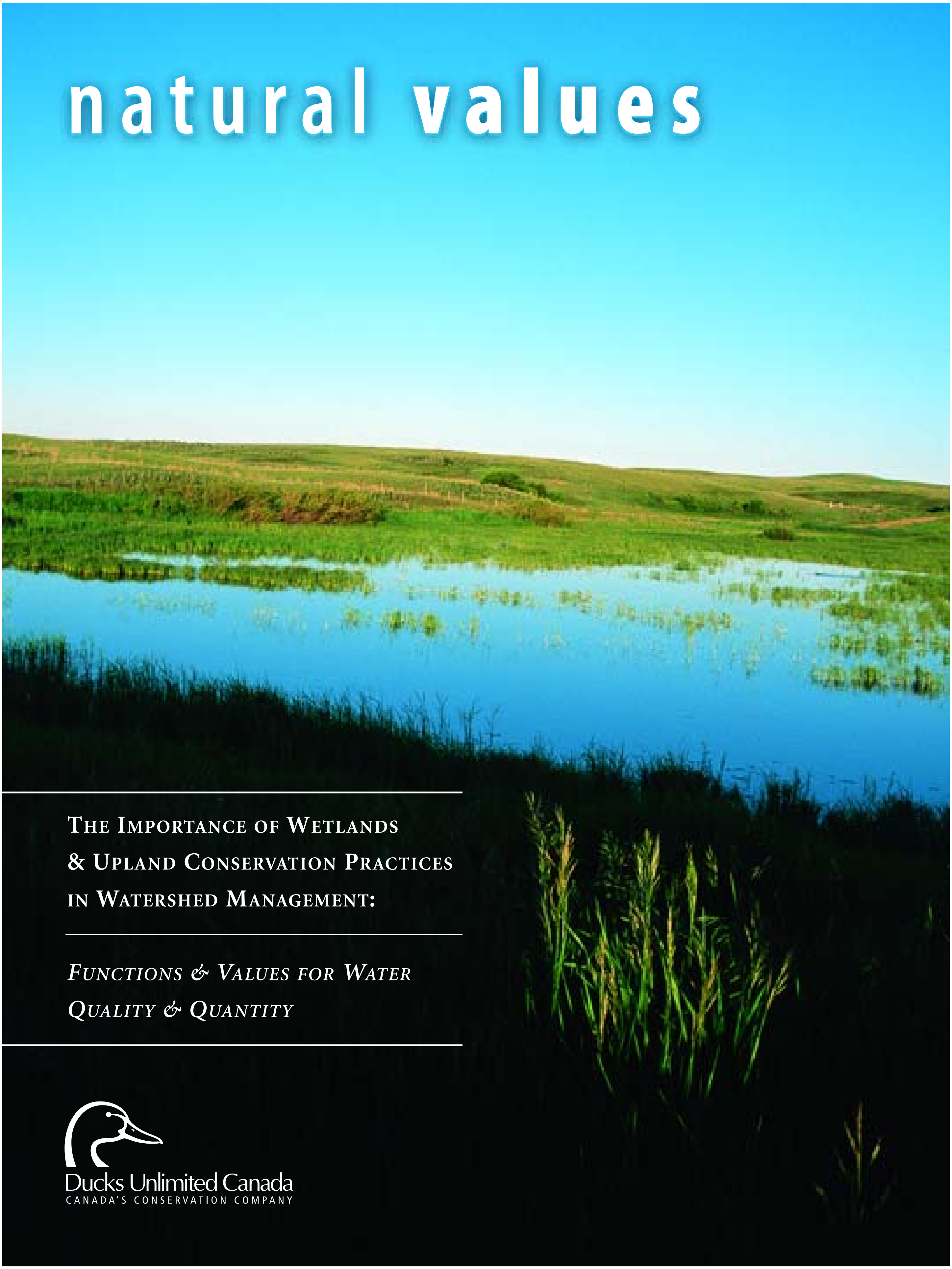

# natural values

## The Importance of Wetlands & Upland Conservation Practices in Watershed Management:

### *Functions & Values for Water Quality & Quantity*

Ducks Unlimited Canada
CANADA'S CONSERVATION COMPANY

# natural values

## The Importance of Wetlands and Upland Conservation Practices in Watershed Management:

### *Functions and Values for Water Quality and Quantity*

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# i executive summary

Fresh water is a vital resource for human society. To ensure the long-term sustainability of water resources, we must protect and enhance landscape features that ensure water quantity and quality in the future. Wetlands and riparian areas are natural watershed features that are critical for sustainable water resource management.

This paper focuses on the function and value of wetlands, riparian areas, and permanent cover within watersheds. An overview of watershed management is presented including Integrated Watershed Management (IWM) planning and implications of land use for water quality.

### *Watersheds*

A watershed is an ecosystem with complex interacting natural components. Upland plant communities, wetlands, riparian areas, rivers, lakes, and streams are critical natural features that affect surface and ground water quality and quantity within a watershed.

Land use, terrestrial landscape features, and modifications to water regimes have a direct influence on not only surface water bodies but also groundwater. Water quality and quantity in a watershed are inexorably linked to processes that are inherent in the landscape, involving surface water, ground water, biogeochemistry, biota, atmospheric deposition, and sedimentation.

IWM planning is a comprehensive multi-resource management planning process involving all stakeholders within the watershed, who, together as a group, cooperatively work toward identifying the watershed's resource issues and concerns, as well as develop and implement a watershed plan with solutions that are environmentally, socially and economically sustainable. Watershed planning is a cyclical, iterative process involving the following key components: evaluation, planning, implementation, and monitoring.

Given the relationships and interdependencies that exist, a comprehensive, all-inclusive approach to considering the factors affecting water resources within a watershed must be clearly understood and considered.

### *Wetlands*

The hydrological functions of wetlands include storage and eventual release of surface water, recharge of local and regional groundwater supplies, reduction in peak floodwater flows, de-synchronization of flood peaks, and erosion prevention. Position in the landscape, location of the water table, soil permeability, slope, and moisture conditions all influence the ability of wetlands to attenuate floodwaters. Wetland drainage reduces the capability of a watershed to attenuate runoff during flood conditions.

Maintaining and restoring wetlands on the landscape reduces overland flow rates and therefore potential flooding.

Recharge of groundwater is an extremely important function of some wetlands; water percolates slowly from wetlands to aquifers. Interactions between wetlands and local or regional groundwater supplies are complex and site-specific and affected by the position of the wetland with respect to groundwater flow systems, geologic characteristics of the substrate, and climate.

Wetlands are extremely complex systems and several characteristics contribute to their roles as nutrient sinks. They retain nutrients in buried sediments, convert inorganic nutrients to organic biomass, and their shallow water depth maximizes water-soil contact and therefore microbial processing of nutrients and other material in the overlying waters. Wetlands can be effective nitrate sinks in agricultural landscapes (over 80% removal). Phosphorus retention in wetlands can also be significant (up to 94%) and is accomplished through adsorption onto particles, precipitation with metals and incorporation into living biomass.

Wetlands can reduce the impacts of sedimentation on water quality within watersheds. Hydrology is a primary determinant of the sediment-retention capacity of a wetland and controls the source, amount, and spatial and temporal distribution of sediment inputs. Percent of wetland area and position are important for reducing sediment loads of water passing through the system.

Little information exists on the effects of the ability of natural wetlands to reduce microbial populations in water. The effectiveness of constructed wetlands to reduce pathogenic organisms from wastewater is high (up to 99% for coliforms). Natural wetlands are dominated by microbes (bacteria, fungi and algae) and plant life that are important for reducing pathogens.

High levels of biological productivity in wetlands result in dissipation of pesticides due to profuse submersed and emergent plant growth that increases the availability of surface area for pesticide adsorption, plant sequestration, microbial degradation, and exposure from wetlands, primarily due to adsorption to organic matter in sediments and decomposing litter.

### *Riparian Areas*

Vegetated buffer strips can effectively control erosion by forming a physical barrier that slows the surface flow of sediment and debris, by stabilizing wetland edges and stream banks, and by promoting infiltration. The required width of a buffer size is determined by the type of vegetation present; the extent and impact of the adjacent land use; and the functional value of the receiving wetland. Studies have found the bulk of sediment removal occurs in the first few meters of the buffer zone; sediment removal can be 75-97%.

Buffer strips can effectively remove nutrients from surface water flow. The main mechanisms of nitrate removal are by vegetation uptake in the roots and anaerobic microbial denitrification in the saturated zone of the soil. Relatively narrow buffers seem to be very (35-96%) effective in reducing nitrogen.

Phosphorus retention can be effective (27-97%) in buffer strips that contain both woody and herbaceous vegetation, grasses and cropped buffer systems. Buffer strips can trap a significant proportion of pathogens (up to 74% of fecal coliforms); however, remaining levels often exceed minimum drinking water standards. Low soil moisture and high soil temperature substantially decrease survival of total and fecal coliform bacteria. The key process for pesticide retention in buffer strips is infiltration. Grass buffer strips can reduce pesticides by 8-100%.

### *Upland Cover*

Upland conservation programs, such as no-till and permanent perennial cover, slow surface runoff, trap sediments and promote infiltration, consequently reducing the amount of sediments, nutrients and pesticides that enter the watershed.

The most beneficial outcome from implementing practices such as conservation tillage and permanent perennial cover is erosion reduction. Erosion from wind, rain and runoff can be reduced up to 99%

because of greater site stability, infiltration and protection as a result of surface crop residue and perennial vegetation.

Upland cover has shown to be effective in reducing the amounts of nitrogen (up to 90%), phosphorus (up to 91%) and pesticides (up to 100%) in runoff but there is potential for increased leaching through the soil profile to groundwater. Although conservation tillage has not always reduced nutrient and pesticide leaching, this practice is recommended because the benefits outweigh the potential drawbacks. With respect to perennial upland cover, this land has been removed from production, resulting in fewer pesticides and fertilizer nutrients applied and subsequently released. Currently there is insufficient information regarding upland conservation practices and pathogen movement.

Wetlands, riparian buffers and uplands are vital components of watersheds and freshwater resource sustainability in North America. If we desire to understand the role of wetlands, uplands and riparian buffers in maintaining both the quantity and quality of water supplies, we must approach policy, management, and research from a holistic viewpoint, incorporating all components of the watershed.

# ii introduction

Fresh water is a vital resource for human society. We depend upon water for drinking, hydropower, irrigation, cooling, and cleaning; for products such as food, plants, and minerals; and for services such as waste purification, transportation, and recreation (Naiman et al. 1995). Currently, freshwater resources are being depleted and degraded on a global scale; as a result, experts agree that research on freshwater ecosystems is paramount to prevent further losses and degradation.

Proper watershed functioning maintains high quality water supplies. Watersheds collect water as rainfall, snowmelt, and runoff; store it for varying lengths of time; and then discharge it as surface runoff or groundwater flow (Black 1997). Wetlands, streams, lakes, and groundwater are all vital components of watersheds; as such, it is important to understand each of these individual components within the larger context of watershed function.

Although the importance of watersheds in the landscape for human communities was first articulated by American John Wesley Powell in the 19th century (Worster 2002), politicians, community-based councils, and land managers have only recently begun acting on a watershed scale to preserve and protect water supplies (e.g., the City of New York in Foran et al. 2000); for aquatic habitat restoration (Bohn and Kershner 2002); and in response to threatened and depleted water resources. Watershed management is an approach to solving residual water quality problems that have not been mitigated with reductions in point-source pollution. Non-point sources (NPS) of pollutants such as agricultural, logging, mining and urban run-off remain significant contributors to water quality impairment, as they are diffuse in nature and often difficult to identify and prevent. The watershed approach moves away from end-of-pipe treatment and toward watershed-specific identification of water quality and quantity problems and integrated solutions (i.e., land use changes, wetland conservation and restoration, buffer establishment, etc.) to such problems (Foran et al. 2000).

Several jurisdictions across Canada have begun the process of developing water strategies and identifying watershed functions in order to develop management plans to protect and enhance these functions while providing guidance for future development (e.g., Conservation Ontario, Saskatchewan Watershed Authority, Manitoba Water Strategy, Alberta Water For Life). Other jurisdictions have also concluded that watershed management is critical for water resource sustainability. The World Water Council's "Long Term Vision on Water, Life, and the Environment in the 21st Century" was designed to ensure sustainable water use for future generations. This process involved consultation and research on water resources at regional and global scales. Canada's vision is built upon, among other principles, integrated water and land resources management systems (Canadian Vision Consultation 1999). This will involve managing land and water by watershed units. Policymakers recognize that sound management practices implemented at the watershed level will protect ecosystem functions; hence, water resources will be protected as well. Watershed management is already an important component of protecting drinking water in many communities.

Wetlands are a continuum within the landscape and interdependent with other landscape units (Bedford and Preston 1988; Mitsch and Gosselink 2000a); thus, alterations to the landscape affect wetland functions. Current scientific understanding acknowledges that landscape factors (i.e., topography, geology, and landscape configuration) and climate influence wetland functions and diversity (Hill and Devito 1997; Bedford 1999). The landscape mediates delivery of water, minerals, nutrients, sediments and biota to wetlands (Brinson 1993; Bedford 1999); it is these factors that determine wetland functions. For example, modifications to watershed hydrology or changes in land use affect how water, nutrients, sediments, and other pollutants are transported to wetlands and other landscape units (Bedford 1999).

Sedimentation and contaminated water supplies can all be attributed in some way to the mismanagement of the watershed (Black 1997). Changes in wetland and other aquatic system water quality often originate from disturbances to the surrounding landscape (i.e., row crops, livestock production, industry, urbanization). In constructing management approaches that improve watershed health, policy makers must keep in mind the influence that soil and cropping systems have within the watershed. Although wetlands and other aquatic habitats are resilient systems that improve water quality, these disturbances can result in changes to the biological, chemical, and physical properties of these systems (Castelle et al. 1994). Maintaining vegetative cover and buffers within a watershed is an important part of reducing or eliminating the impacts of agricultural and urban land use activities on wetlands and other aquatic systems.

Wetlands, uplands and riparian buffers are vital components of watersheds and freshwater resource sustainability in North America. If we desire to understand the role of wetlands, uplands and riparian buffers in maintaining both the quantity and quality of water supplies, we must approach policy, management, and research from a holistic viewpoint, incorporating all components of the watershed.

### *Objectives*

*The objective of this document is to provide an overview of the importance of watershed management for sustaining freshwater resources and to provide the state of the science with respect to the benefits of wetlands, riparian areas, and upland cover for sustainable water resource management.*

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# watershed management

A watershed encompasses an area of land that drains to a single body of water, whether a river, wetland or estuary, and may range in size from a few hundred hectares to a catchment basin the size of the Mississippi River's, which drains roughly 40 percent of the U.S.A. (Worster 2002).

A watershed is an ecosystem with complex interacting natural components. Upland plant communities, wetlands, riparian areas, rivers, lakes, and streams are critical natural features that affect surface and ground water quality and quantity within a watershed. The functions and values provided by natural features must be included in the development of a watershed management plan. The conservation of wetlands and riparian areas is a fundamental building block of any watershed plan.

Land use, terrestrial landscape features, and modifications to water regimes have a direct influence on not only surface water bodies (Baron et al. 2002) but also groundwater. Water quality and quantity in a watershed are inexorably linked to processes that are inherent in the landscape, involving surface water, ground water, biogeochemistry, biota, atmospheric deposition, and sedimentation (Swanson et al. 2003). In particular, the hydrologic cycle has a strong influence on nutrient, chemical and sediment transport processes. An understanding of all components of a water budget is desirable although not always practical. Winter (2001) suggests the use of fundamental hydrologic landscape units to describe the movement of surface and ground water, how they interact, and how they are affected by climate. Hydrologic landscapes are described in terms of land-surface form, geologic framework, and climatic setting, and may be a foundation for uniform conceptualization of actual landscapes.

Gaps still exist between watershed management and basic knowledge of hydrological and ecological processes. As such, modeling may aid decision makers in considering multiple environmental objectives for management on a watershed scale (Newbold 2002). For example, an integrated watershed management framework combining economic, environmental, and GIS modeling was developed by Yang et al. (2001a, 2001b) to study cost-effective land retirement under the Conservation Reserve Enhancement Program (CREP) in the United States. They applied the framework to 12 agricultural watersheds in the Illinois CREP region under two standards: (i) a uniform standard under which each watershed is required to achieve the same sediment reduction goal; and, (ii) a non-uniform standard under which the marginal cost of sediment abatement is equal across watersheds. They found that the cost of abatement under the uniform policy are 38% higher than under the marginal cost policy, suggesting that the non-uniform standard be modified to allow sloping cropland adjacent to streams to be eligible for registration in CREP to improve cost-effectiveness.

*Integrated Watershed Management*

Integrated Watershed Management (IWM) planning is a comprehensive multi-resource management planning process involving all stakeholders within the watershed, who, together as a group, cooperatively work toward identifying the watershed's resource issues and concerns, as well as develop and implement a watershed plan with solutions that are environmentally, socially and economically sustainable.

In order that IWM be effective, plans must develop within a framework that outlines the vision for the watershed, principles, processes, goals and expected outcomes. Watershed planning is a cyclical, iterative process involving the following key components: evaluation, planning, implementation (i.e. act), and monitoring.

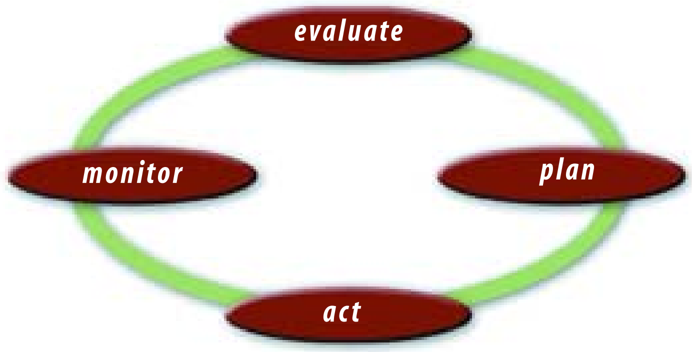


**FIGURE 1 –** The watershed planning process; evaluation, planning, implementation, and monitoring.

IWM acknowledges that nothing happens in isolation and that everything is connected by the land and water within the watershed. Given the relationships and interdependencies that exist, a comprehensive, all-inclusive approach to considering the factors affecting water resources within a watershed must be clearly understood and considered. Recognition and acknowledgement of the linkages and relationships between other resources such as agriculture, forestry, wildlife, waterfowl and fish with water must be part of the integrated approach to watershed management.

### *a) Implications of Land Use for Water Quality*

Landscape characteristics (land use, geomorphology, climate conditions) have been linked to water quality, water quantity, and run-off in numerous studies encompassing varying spatial scales. Catchment-scale studies have become increasingly more common; however, a large investment of time and resources is often required as a unique combination of characteristics influence water quality within each watershed (Sliva and Williams 2001).

Spatial analysis of water quality and land use/land cover have been used to evaluate watershed health and to develop models for assessment of potential nutrient and sediment loadings based on land use. Landscape metrics (i.e., urban, agriculture, forest, wetlands, or barren classifications) explained high percentages of variation in nitrogen (65 – 86%), dissolved phosphorus (73%), and sediment (79%) loading to streams in the Chesapeake Bay basin (Bruce Jones et al. 2001). GIS and multivariate analysis were used to determine correlations between water quality and landscape characteristics in three southern Ontario watersheds (Sliva and Williams 2001). Urban land use was found to influence water quality the most strongly, the influence of agricultural land was variable, and forested areas were important in mitigating water quality degradation. The authors suggested that "secondary" data sources (governmental water quality and quantity data), used in their study, are not sufficient in scale or accuracy for detailed analysis of water quality – land use relationships; however, they are suitable for hypothesis generation and identification of data gaps. Tong and Chen (2002) linked regional land use in Ohio to in-stream nitrogen, phosphorus and fecal coliform bacteria using a combination of GIS and hydrologic modeling.

The impacts on water quality of agricultural land uses in particular have been identified in numerous studies at the watershed scale. Berka et al. (2001) linked water quality (nitrate, ammonia, phosphorus, and coliform levels) with agricultural intensification in a rural watershed in the Fraser River lowland in British Columbia and Washington State. Crop management systems in Illinois were linked to nitrate concentrations in tile drains and surface streams (Mitchell et al. 2000), and high levels of fecal coliform were observed downstream of livestock and septic systems in a rural Nova Scotia watershed (Jamieson et al. 2003). Surface water quality in the Upper Oconee watershed in Georgia was related to land use (Fisher et al. 2000). In particular, phosphorus, nitrogen, and fecal coliform bacteria were high near intensive poultry production in the headwaters, but were reduced within the watershed prior to reaching the intake for a municipal water supply. Their analysis identified areas that should be priorities for natural resource management to reduce agricultural NPS pollution.

Smaller-scale studies have also linked non-point source (NPS) pollutants in surface and groundwater to conventional agriculture. Kemp and Dodds (2001) found that groundwater nitrate concentrations in Kansas were highest near intensive agriculture and lowest in upland, pristine tallgrass prairie sites. Conversion of an existing agricultural system to sustainable land use practises (buffer strips, fallow strips, minimum tillage, organic farming, etc.) resulted in reduced surface and ground water loads of nitrogen and phosphorus in Germany (Honisch et al. 2002).

### *b) Watershed Processes*

Watershed-scale assessment of the benefits of riparian areas, prairie restoration, agroforestry and other conservation measures for reducing NPS pollution from agricultural landscapes has been undertaken at various spatial scales (5 ha to 5000 ha) using paired watershed experimental design. Riparian restoration, livestock exclusion and streambank protection in two treatment watersheds (approximately 1000 ha in size) were evaluated with respect to phosphorus loading in the Lake Champlain basin in Vermont (Meals and Hopkins 2002). Statistical calibration between the treatment watersheds and the control watershed (no modifications to grazing land) was achieved over three years prior to treatment. Two years of post-treatment data documented significant reductions in phosphorus concentrations and loads from both treatment watersheds (Table 1).

Schilling (2002) compared nitrate, Cl and $SO_4$ loads from paired 5000 ha watersheds in Iowa: Walnut Creek, during conversion from row crop agriculture to native prairie, and Squaw Creek, a highly agricultural control watershed. Average nitrate and Cl loads were significantly lower in Walnut Creek than Squaw Creek (Table 1). Udawatta et al. (2002) compared runoff, sediment, and nutrient losses from three small watersheds in Missouri after a six year calibration and three year treatment period. Treatments consisted of agroforestry (mix of trees and vegetated buffer strips), contour strips (grass buffer strips), and a control (corn-soybean row crops). After three years of post-treatment monitoring, both contour strip and agroforestry management reduced NPS pollution in runoff (Table 1).

**TABLE 1 – Range of percent reduction in sediment, nitrogen, and phosphorus in restored agricultural watersheds.**

| | Treatment | Reduction (%) |
|---|---|---|
| Nitrate load | restored prairie | 30 - 50 |
| Phosphorus concentration | riparian restoration | 21 |
| Phosphorus load | riparian restoration | 21 - 41 |
| Phosphorus load | agroforestry/contour strips | 8 - 17 |
| Nitrogen load | agroforestry/contour strips | 20 - 21 |
| Nitrate load | agroforestry/contour strips | 24 - 37 |
| Sediment load | agroforestry/contour strips | 0 - 19 |

### c) Case Studies

Watershed management has become a priority in large basins that contribute to eutrophication, pollution, and hypoxia of coastal waters such as the Gulf of Mexico, Chesapeake Bay, and the Neuse River Estuary. Nitrogen loading in the Mississippi River Basin has been linked to an extensive hypoxic zone (i.e., low levels of dissolved oxygen in bottom waters) in the Gulf of Mexico, which may impact the health of aquatic biota (see a review by Mitsch et al. 2001). A similar situation in the Neuse River estuary caused extensive fish kills in 1996, leading to the implementation of a nutrient management strategy in the Neuse River Basin (Wossink and Osmond 2002). Agricultural nutrient inputs to Chesapeake Bay have led to increased phytoplankton productivity and hypoxic waters (Woltemade 2000).

The efforts to reduce damaging effects on estuaries, bays, and other coastal waters typically focus on watershed-scale reduction of NPS nutrients through restoration of wetlands, use of riparian buffers, and other conservation measures (e.g., Mitsch et al. 2001, Tetra Tech Inc. 2003). Mitsch et al. (2001) suggest that nitrogen loading from the Mississippi River Basin to the Gulf of Mexico could be reduced by 40% using a combination of techniques, including creation and restoration of wetlands and riparian buffers, modification of farm practices, and flood control. Changing farm practices and using wetlands and riparian buffers had much greater potential for N reduction than flood control or reducing point sources (i.e., 300 – 1400 vs. 20 – 100 thousand metric tons per year). An economic analysis of the cost-benefit ratio of reducing N loading in the Mississippi Basin by changing the way nutrients are managed on the field vs. intercepting nutrient-rich runoff with restored wetlands suggested that cost-effectiveness depended on the level of N reduction desired (Ribaudo et al. 2001). Wetland restoration was more cost-effective than managing fertilizer application for achieving 26% or greater N reduction. Natural nitrogen sinks (i.e., wetlands and riparian buffers) are effective and should be restored or placed between agricultural fields and streams and rivers of the Mississippi River Basin.

Watershed management approaches have been adopted by many jurisdictions to protect not only ecological integrity but also drinking water supplies, including New York City and the Neuse River Basin in North Carolina. The City of New York implemented enhanced watershed regulations in 1997 with the intent to manage and protect the water supply of over 8 million people (NYC 2003). New York City planners have developed a watershed protection plan as an alternative to the future need for a water filtration plant (Ehlers et al. 2000). Partnership programs targeting specific sources of pollution in the watershed are one of three key programs implemented in the watershed protection plan, in addition to acquisition of watershed lands and enforcement of watershed regulations. A large effort has been undertaken to develop agricultural Beneficial Management Practices (BMPs), including taking environmentally sensitive lands out of production (under the Conservation Reserve Enhancement Program). Other partnership programs have included remediation of failing septic systems, storage of winter road de-icing materials, and BMPs to address stormwater runoff (NYC 2003). The Upper Neuse Watershed Management Plan was developed in response to concerns about the long-term sustainability of water supply quality and stream habitat in the Upper Neuse river basin in response to projected population growth in the basin (Tetra Tech 2003). Management techniques were recommended to address problems of nutrients, algae, sedimentation, and erosion in the basin, including restoration of stream buffers and wetlands to restore natural watershed functions. In addition to new development site controls to reduce runoff and nutrient loads, public education and stewardship, monitoring and enforcement, and point source control, the techniques are intended to be employed together to mitigate and prevent pollution within the Upper Neuse river basin.

The following sections outline the current science relating to wetlands, riparian buffers, and upland conservation practices that provide water quality and quantity benefits within a watershed.

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# wetlands

### a) Overview of Wetlands

There are five classes of wetlands recognized in Canada: bog, fen, swamp, marsh, and shallow water (National Wetlands Working Group 1997). This classification system recognizes that hydrological processes dictated by climate and landscape factors largely determine wetland form (National Wetlands Working Group 1997).

Wetlands can also be evaluated according to their position in the landscape, or site type. Four different types are recognized: lacustrine, riverine, palustrine, and isolated (OMNR 1993). Lacustrine wetlands are associated with lakes. They can occur at the mouth of a river, at the shoreline of the lake but separated from the lake by a barrier beach, or exposed to the lake. Riverine wetlands are adjacent to streams and rivers. They may be located within the channel, adjacent to the stream, or on the flood plain. Palustrine wetlands occur upslope of riverine or lacustrine wetlands. They may or may not have an inflow, and have intermittent or permanent outflow. An isolated wetland receives nutrients from precipitation, overland flow, and groundwater.

Our understanding of wetland development has evolved since the initial classification scheme was adopted. Physical and chemical factors are presently thought to interact with biological processes to determine wetland characteristics (Winter and Woo 1990; Bedford 1999; Winter 1999; Price and Waddington 2000). For example, there is now emphasis on the influence of hydrology, topographic location, thickness and permeability of soils, underlying geological characteristics, regional climate, and other landscape characteristics on wetland functions (Winter and Woo 1990; Brinson 1993; Hill and Devito 1997; Bedford 1999; Winter 1999; Price and Waddington 2000). As such, the U.S. Army Corps of Engineers has promoted the hydrogeomorphic (HGM) approach to wetland classification and assessment (Brinson 1993, 1995, 1996 in Cole et al. 2002). This classification focuses on position in the landscape and hydrology (e.g., slope, headwater floodplain, mainstem floodplain, etc.); however, because regional differences in soils, climate, etc. can affect wetland functions, caution is advised when attempting to apply models of classification between regions (Cole et al. 2002).

Many wetlands occur in topographic depressions created by glacial erosion and deposition (Winter and Woo 1990). Wetlands can intercept the water table in such a way that they have only inflows and no outflows (Figure 2-a) (Mitsch and Gosselink 2000, 135). Other wetlands occur in areas of steep land slopes such as embankments or river valley walls where groundwater discharges directly to the land surface from the underlying soil or emerges from surrounding uplands creating an area of permanently saturated soil (i.e., discharge wetland)(Figure 2-b)(Hill 1990; Roulet 1990; Winter and Woo 1990, Mitsch and Gosselink 2000, 135). This occurs when the water level in the wetland is lower than the water table of the surrounding land. This type of wetland can be an isolated low point in the landscape, but most often it discharges excess water downstream as surface water or groundwater (Mitsch and Gosselink 2000, 135)(Figure 2-c). When the water level in a wetland is higher than the water table, groundwater will flow downward from the wetland to underlying water-saturated soil (i.e., recharge wetland)(Figure 2-d). When a wetland is above the groundwater of an area the wetland is referred to as being perched and loses water through infiltration into the ground and through evapotranspiration (Figure 2-e).

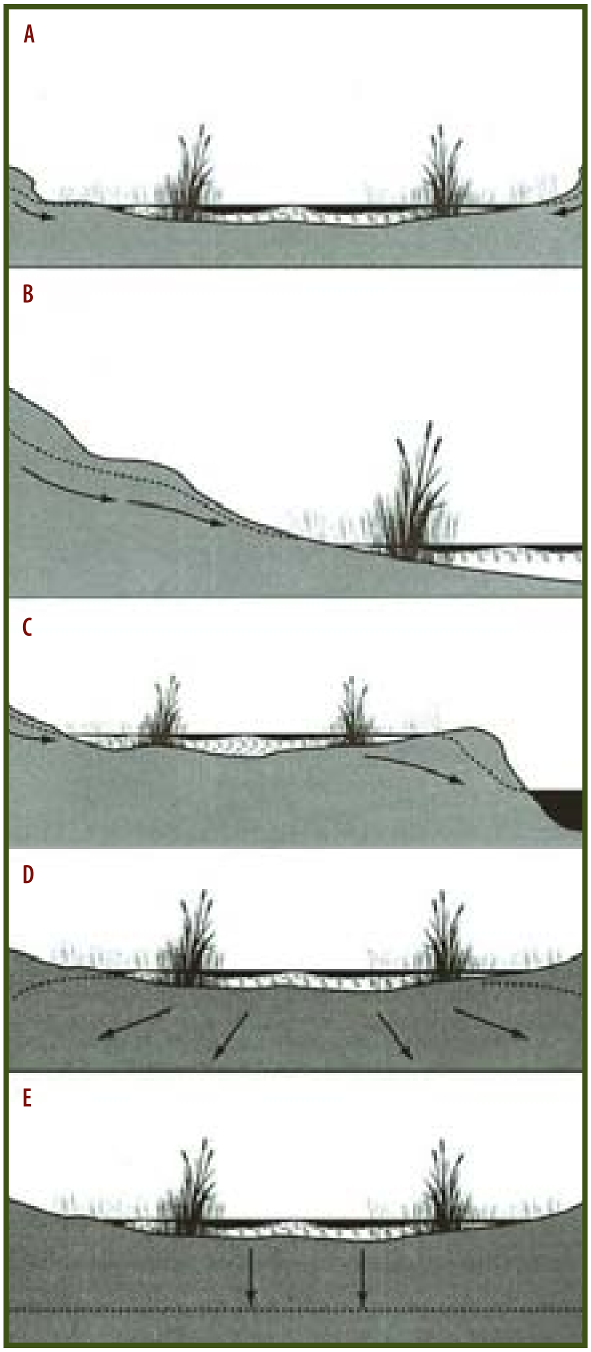


**FIGURE 2 –** **Possible discharge – recharge interchanges between wetlands and groundwater systems including: (A) marsh as a depression receiving groundwater flow; (B) groundwater spring or seep wetland or groundwater slope wetland at base of a steep slope; (C) floodplain wetland fed by groundwater; (D) marsh as a recharge wetland adding water to groundwater; (E) perched wetland or surface water depression wetland (Mitsch and Gosselink 2000).**

### *b) Hydrological Functions of Wetlands*

The hydrological functions of wetlands include storage and eventual release of surface water, recharge of local and regional groundwater supplies, reduction in peak floodwater flows, de-synchronization of flood peaks, and erosion prevention (Carter 1986; LaBaugh 1986; Winter and Woo 1990; LaBaugh et al. 1998; Winter 1999; Mitsch and Gosselink 2000a; Price and Waddington 2000, Meyer et al. 2003). Each situation is unique and dependent on local topography, climate, geology, and watershed characteristics (e.g., LaBaugh et al. 2000). Land use is also a key factor influencing wetland hydrology. For example, the conversion of cultivated land to brome grass surrounding a prairie wetland resulted in enhanced trapping of snow and infiltration into frozen soil (van der Kamp et al. 2003). In a review of hydrological processes in temperate headwater wetlands in the glaciated regions of northeast North America, Taylor (1997) concluded that wetland hydrology is complex, depending on wetland type, catchment characterisitics, and climatic conditions.

In order to understand the hydrological functions of wetlands, it is necessary to have a working knowledge of wetland hydrology. Wetlands are dynamic, continuously receiving and losing water through interchange with the atmosphere, surface flow and groundwater (Winter and Woo 1990). Water source to wetlands is highly variable, ranging from almost completely precipitation-derived to groundwater-sourced (Winter et al. 2001). Although significant advances have been made in our understanding of wetland hydrology in recent years (Hill and Devito 1997; Winter 1999; van der Kamp et al. 2003) we have a limited understanding of wetland hydrology for the wide variety of wetland types that exist. This ultimately affects our understanding of many wetland functions, as water is the primary agent of material and nutrient transfer in and out of wetlands (Doss 1995, Hill and Devito 1997, Hill 2000). Many non-hydrological functions of wetlands depend on hydrology. For example, biogeochemistry in several Ontario wetlands and streams has been linked to hydrology (Hill 1990; Hill 1996; Devito et al. 2000a). Ultimately, hydrological characteristics that influence wetland chemistry are a function of climate and landscape features such as depth of permeable sediments, groundwater flow patterns, organic deposits, and geology (Hill 1996; Brinson 1993; Bedford 1999; Winter 1999; Devito et al. 2000a).

A wetland water budget is an equation in which the inputs, outputs, and change in storage of water in the wetland are balanced.

**(Equation 1)** $$P + SWI + GWI = ET + SWO + GWO + S$$

**Where P = precipitation, SWI = surface water inflow, GWI = groundwater inflow, ET = evapotranspiration, SWO = surface water outflow, GWO = groundwater outflow, S = change in storage**
*(Carter 1986)*

Each component of the water budget can be complicated to measure and incomplete characterization of wetland hydrology is often the result of accumulated errors inherent in measuring each of these components of the water budget equation. Observation and measurement of runoff and other components of the water budget typically are difficult, further complicating study of wetland hydrology.

*1. Water Storage and Flood Reduction*

Flood reduction is an important wetland function, both environmentally and economically. Flooding causes undesirable effects downstream, such as erosion of shorelines and riverbanks, erosion of agricultural soil (by overland flooding), sedimentation of eroded soil further downstream, washout or siltation of fish spawning areas, and damage to homes and businesses. The ability of wetlands to store incoming water is highly variable. Position in the landscape, location of the water table, soil permeability, slope, and antecedent moisture conditions influence the ability of any given wetland to attenuate floodwaters (Carter 1986; Winter and Woo 1990; Devito et al. 1996; Cey et al. 1998).

Wetlands commonly retain surface inflow as the basin fills and then release the accumulated water during an extended period (Winter and Woo 1990). The degree of flow modification depends on the characteristics of the wetland basin and the timing and magnitude of flow. Where streams enter the wetland and then reappear at the lower elevation of the wetland outlet, there is a thorough mixing of surface and subsurface water and the flow pattern is greatly modified (Winter and Woo 1990). Wetland vegetation slows water flow significantly (Carter et al. 1978). As surface water enters a wetland, vegetation disperses incoming water, reduces flow velocity, and thus increases residence time of water in the wetland (Brown 1988). Streams flowing through a wetland along well-defined channels have less exchange with groundwater and the stream flow regime is little changed by the wetland (Woo and Valverde 1981).

Prior to drainage of productive land on the Prairies, the numerous small depressions in morainal areas (i.e., that are not part of an integrated drainage system) only rarely contributed to stream flow (Winter 1989). Typically the water would run into depressions and infiltrate during the frost-free period. Hubbard and Linder (1986) suggested that numerous small wetlands in the prairie pothole region cumulatively store a large amount of spring runoff, based on the extrapolation of data collected from 213 wetlands on 648 ha. Hayashi et al. (2003) studied small ($<1000\ m^2$) depressions (i.e. wetlands) and found that they stored a large portion, or all, of the snowmelt runoff generated in their respective watersheds. Although each wetland has a small storage, they collectively provide a significant storage capacity.

Wetlands may play a key role in controlling stormwater runoff; however, the flood reduction benefits of wetlands are often seasonal. Wetland water storage occurs underground in saturated soils or in surface depressions (Winter and Woo 1990). When water tables are low during the dry season, considerable storage capacity is available in unsaturated peat and wetlands are effective in retarding or preventing runoff. Wetlands that are saturated may have little capacity to store water and any additional water may run off the wetland quickly (Verry and Boelter 1979; Winter and Woo 1990; Devito et al. 1996). For example, Taylor (1982) showed that small wetlands near Peterborough, Ontario held back runoff in the summer months, when water levels were low. However, in the spring and fall, storage capacity was exceeded and runoff was released downstream.

Hydrological models, spatial analysis, and computer simulations have been used to demonstrate the ability of wetlands to store surface runoff. Bertulli (1981) simulated a flood on the Napanee River, Ontario under two scenarios: one with the existing wetland in place, and one without the wetland. The computer-simulated flood hydrograph showed that the presence of the wetland would reduce peak discharge from 150 cubic meters per second ($m^3/s$) to 80 $m^3/s$ by extending the period of time over which the floodwaters moved through the river. Ludden et al. (1983) estimated the runoff storage capacity of wetland areas in the Devils Lake basin in North Dakota. They calculated that approximately 72% of the total runoff from a rain event with a two-year frequency, and 41% of the runoff from a rain event with a 100-year frequency, would be retained by these wetland depressions. Spring runoff retention in boreal forest peatlands in northern Manitoba is influenced by frost-table dynamics and surface storage conditions (i.e., water levels), and therefore interannual variation is common (Metcalfe and Buttle 2001).

Positive benefits of maintaining wetlands in the landscape are well known. The United States (U.S.) Army Corps of Engineers recommended the acquisition and protection of wetland areas along the Charles River in Massachusetts as the least expensive method of flood control (Carter et al. 1978). Miller and Nudds (1996) linked the large 1993 and 1995 floods in the Mississippi River Valley to wetland drainage, and demonstrated that wetland drainage in the U.S. is correlated with greater river flow rates than in Canada, where landscape alteration is much less severe. Hey and Philippi (1995) suggested that the restoration of approximately 5.3 million ha in the Upper Mississippi and Missouri Basins would provide enough floodwater storage (approximately 1m deep) to accommodate excess river flows associated with the 1993 flooding in the U.S. midwest. They concluded that restoration of an estimated 7% of the watershed would be sufficient to deal with even extreme event floods on a large scale.

Wetland modification may be equally detrimental as wetland loss to a watershed's runoff-holding capacity. For example, wetland channelization, which often occurs in urban areas, leads to increased runoff and loading from a basin. Whiteley and Irwin (1986) found that, of two creeks flowing into the Beverly Swamp in Ontario, the unchannelized stream delayed flood peaks by 20 to 30 hours and reduced peak flows compared to the stream with a well-defined channel. Brown (1988) found that stormwater runoff from Lamberts Creek, Minnesota was highest in subwatersheds with channelized wetlands and steep slopes. The subwatershed with a large percentage of unmodified wetlands (94%) exhibited a long steady storm discharge ($<0.5$ $m^3/s$ over 24 hours for a June storm) and low total runoff volume (0.01 x 106 $m^3/km^2$ for the 12 storms sampled). The sub-watershed with the highest degree of urban land use and wetland channelization had large peaks of discharge during storms (2.5 $m^3/s$ over 2-3 hours), and the greatest total runoff (0.14 x 106 $m^3/km^2$).

Research conducted in the Oak Ridges Moraine of Ontario has indicated that wetlands located on this unique geologic formation may not function in flood reduction; instead, they may actually be sources of rapid overland flow (e.g., Hill and Waddington 1993; Waddington et al. 1993; Cey et al. 1998). This is attributable to high water tables and groundwater discharge (95% of annual inputs come from underlying aquifer) to these wetlands. Recent efforts to model groundwater and surface water mixing during storm events have been used to attempt to explain this phenomenon (e.g., Brassard et al. 2001).

*2. Groundwater Recharge*

Recharge of groundwater is an extremely important function of some wetlands and occurs when water percolates slowly from wetlands to underground aquifers. Interactions between wetlands and local or regional groundwater supplies are complex and site-specific (Hill 1990; Winter and Woo 1990; Winter 1999, Devito et al. 2000a; Price and Waddington 2000). The interaction between wetlands and groundwater is affected by the position of the wetland with respect to groundwater flow systems, geologic characteristic of the substrate and climatic setting (Winter 1999). A wetland can recharge groundwater supplies, or be a site of groundwater discharge (Carter 1986; Hill 1990; LaBaugh et al. 1998) (Figure 3).

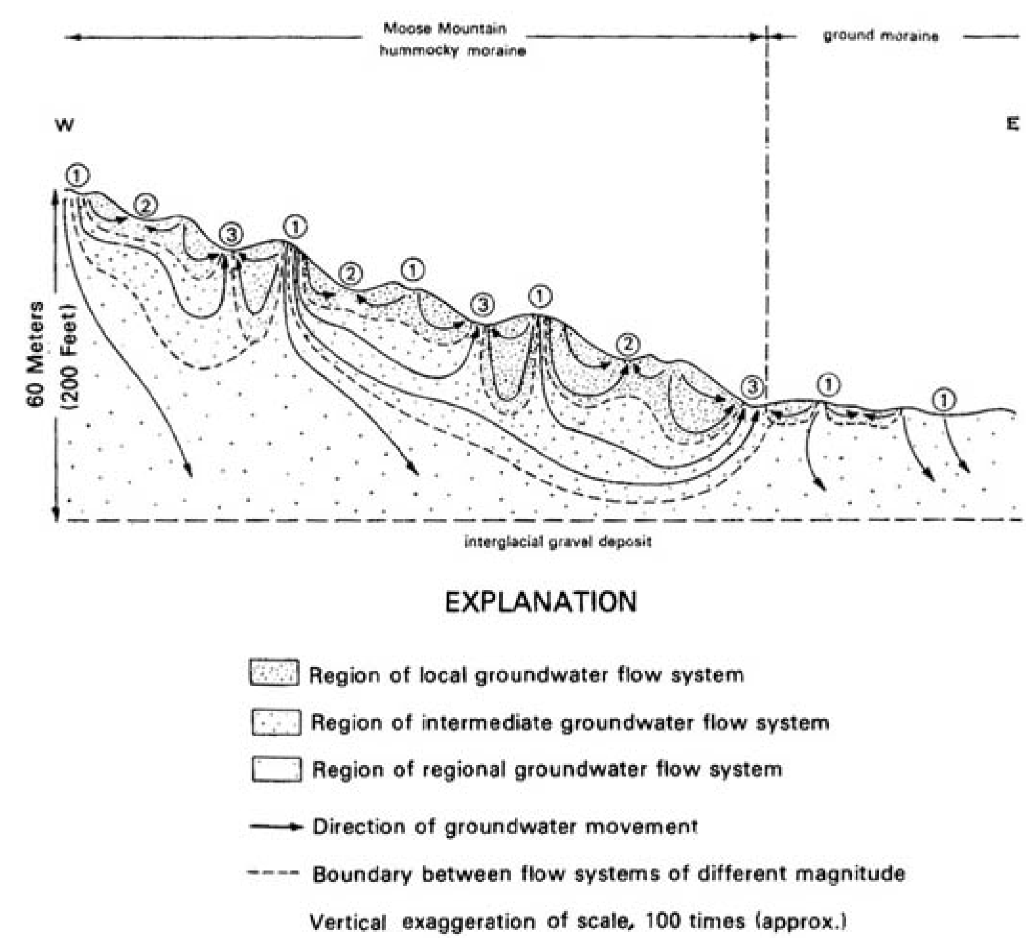


**FIGURE 3 – Diagrammatic section of the flow systems of Moose Mountain, Saskatchewan (1) recharge areas, (2) discharge area of local flow system, (3) discharge area of local and intermediate flow systems (Winter 1989).**

Groundwater recharge occurs from many areas in the landscape, including wetlands (from seasonal to permanent) and uplands (Winter 1988; van der Kamp and Hayashi 1998). Permeability of surficial sediments and geologic formations is referred to as hydraulic conductivity, and is a measurement of the ability of water to move through a specific type of soil or deposit (van der Kamp and Hayashi 1998). Hydraulic conductivity of materials overlying aquifers may determine the rate of aquifer recharge. In general, glacial drift in eastern North America is permeable because the bedrock from which the drift is derived is more permeable than in other areas (Winter and Woo 1990).

Prairie potholes in the semi-arid portion of the northern prairies are known to be important for groundwater recharge, despite low hydraulic conductivity of clay-rich glacial deposits (van der Kamp and Hayashi 1998). The ponded water in wetlands is generally connected to and continuous with the water table in the surrounding area (Figure 3) and therefore all seepage of water from the ponds into the subsurface can be regarded as groundwater recharge. Recharge to the aquifers depends on the availability of water in wetlands and other depressions in the overlying landscape, as well as the hydraulic conductivity of overlying aquitards. van der Kamp and Hayashi (1998) found that estimates of recharge from prairie potholes (2-45 mm/yr) were similar to published rates of aquifer recharge in other areas (5-40 mm/yr), suggesting that these wetlands are the main source of recharge to regional aquifers. Since then, other studies have shown that small wetland depressions are important in both groundwater recharge and water storage in many physiographic settings, including the northern glaciated prairies (Hayashi et al. 2003) and Appalachian Ridge and Valley province (Moorhead 2001). Hayashi et al. (2003) used piezometers and infiltrometers to study infiltration of snowmelt water under small depressions in Saskatchewan. Water table wells and piezometers were used to study seasonal patterns of the water table in an Appalachian mountain fen, and indicated that the fen is an aquifer recharge area (Moorhead 2001).

Hydrology studies in eastern Canada have shown that groundwater recharge by wetlands is variable. Gehrels and Mulamoottil (1990) completed a comprehensive water budget for the Hidden Valley wetland in Kitchener, Ontario. They discovered areas of both groundwater discharge and recharge within the same wetland, confirming the often complex nature of wetland hydrology. Groundwater accounted for 36% of all water flowing into the wetland and 53% of all water discharging from the wetland. Whiteley and Irwin (1986) reviewed a study of the Beverly Swamp north of Hamilton, in which the authors found that of the two streams that enter the swamp, one recharged groundwater from June to September and the other continually received groundwater discharge. In another study reviewed by Whiteley and Irwin (1986), the authors found that the Telford peatland in southern Ontario recharged the regional watertable, with seepage of up to 135 mm. Research on the Oak Ridges moraine in southern Ontario conducted by Hill (1990) and Hill and Devito (1997) show that some wetlands in the region do not provide recharge to aquifers but receive significant groundwater discharge from an aquifer. For example, Mill Creek in Ontario is sustained by groundwater discharge – rainwater recharges groundwater, which then flows into the valley, forms wetlands, and ultimately discharges into Mill Creek. This clean, cold water is crucial for the existence of a cold water fishery in the stream (Grand River Conservation Authority 1997).

Although significant advances have been made in our understanding of wetland hydrology (Winter and Woo 1990; Winter 1999; Price and Waddington 2000) there is a definite need for more information on the factors influencing the hydrological functions of wetlands. Winter (1999) outlines the complexity of groundwater recharge and discharge by stating that streams, lakes and wetlands are integral parts of groundwater flow systems. Fluxes of water to and from groundwater reflect the positions of the surface-water bodies with respect to different-scale groundwater flow systems; local geologic control of seepage distribution through their beds, and the magnitude of transpiration directly from groundwater around their perimeters, which intercepts potential groundwater inflow or draws water from the surface-water body. Understanding the relative importance of all these factors for a given water body is needed for integrated water resource management (Winter 1999).

### *c) Water Quality Functions*

Wetlands influence many aspects of water quality, including nutrients, suspended solids, pathogenic microbes, and anthropogenic pollutants such as pesticides. Because of their high biological productivity, wetlands can transform many pollutants into harmless byproducts via natural processes (Kadlec and Knight 1996). This makes them ideal for processing wastewater, and as a result, constructed wetlands have become common for primary, secondary, and tertiary treatment of sewage. Information on the design and performance of constructed wetlands for improving water quality is readily available for the U.S. (e.g., Kadlec and Knight 1996), and to some degree for Canada (see Kennedy and Mayer 2002). Constructed wetlands efficiently remove total nitrogen and phosphorus (Kirby 2002), BOD (Knowlton et al. 2002), and fecal coliform (Knowlton et al. 2002, Hill and Sobsey 2001, Kern et al. 2000) from municipal and animal production facility wastewaters (Table 2). Constructed wetlands are also used to treat a wide range of surface waters, nonpoint source pollutants in runoff, and domestic and industrial effluents. Examples in the recent literature include treatment of lake water from Lake Apopka, FL (Coveney et al. 2002) and septic tank effluents in Ohio (Steer et al. 2002). Although there is evidence of significant improvement of effluent quality by constructed wetlands (Table 2), site-specific conditions may prevent some parameters from meeting applicable guidelines for receiving waters (e.g., Hench et al. 2003, Steer et al. 2002).

Natural wetlands have been the subject of much investigation with respect to water quality functions. Early studies focused on the effects of a wetland's position in the landscape on downstream water quality (e.g. Whigham et al. 1988; Johnston et al. 1990; Detenbeck et al. 1993; Weller et al. 1996). Debate about whether wetlands located further upstream within a watershed relative to others have a greater impact on water quality and flood protection is ongoing (DeLaney 1995); however, there is evidence that the greater the wetland area, the greater the benefits. For example, Detenbeck et al. (1993) evaluated the effect of "wetland mosaics" on surface water quality of 33 lakes in Minnesota. They derived 27 variables using Geographical Information Systems (GIS) to describe land use, soils, topography, and wetlands, and found that wetland area, agriculture land use, urban land use, herbaceous wetlands, and forest described most (85%) of the variance in surface water quality (nutrients and suspended solids). They concluded that water quality is high in lakes with nearby wetlands, and in lakes with forested watersheds. Johnston et al. (1990) conducted a similar study on the effect of wetlands on stream water quality, and again found that water quality was correlated with the proximity of wetlands. Conversely, Devito et al. (2000b) found that total phosphorus (TP) in boreal lakes was higher in those with larger areas of surrounding wetlands area due to near-surface hydrologic flushing to the lake. Landscape processes are variable and produce site-specific biogeochemical functions (Hill and Devito 1997).

**TABLE 2 – Percent reduction in total nitrogen (TN), nitrate-N ($NO_3$), ammonia-N ($NH_4$), total phosphorus (TP), phosphate (SRP), sediment (TSS) and pathogens in constructed wetlands.**

| Author | Location | TN | $NO_3$ | $NH_4$ | TP | SRP | TSS | Pathogens |
|---|---|---|---|---|---|---|---|---|
| Coveney et al. (2002) | Florida; constructed wetland treating lake water | 30-52 | ++ | ++ | 30-67 | ++ | 89-99 | |
| Falabi et al. (2002) | constructed Lemna pond | | | | | | | 61-62[a] 89-98[b] |
| Hill and Sobsey (2001) | North Carolina; constructed wetland receiving swine effluent | | | | | | | >96[c] |
| Kern et al. (2000) | Germany; constructed wetland receiving swine effluent | | | | | | | 95[d] |
| Kirby (2002) | Nova Scotia; 2 constructed wetland receiving municipal effluent | 77, 87 | | | 83, 90 | | 45, 48 | 98, 99.8[d] |
| Knowlton et al. (2002) | Missouri; constructed cattail wetland | 36 | | 17 | 4 | | | 97[d] |

*++ soluble nutrients increased* **a** *Total and fecal coliform* **b** *Cryptosporidium and Giardia* **c** *Fecal coliform, Salmonella, E. coli* **d** *Fecal coliform*

To determine the effectiveness of wetlands for improving water quality it is important to have an in-depth understanding of wetland nutrient cycling. Often, wetland water quality studies focus only on the chemical concentration of water as it enters and leaves the wetland (Kadlec and Kadlec 1978). In these studies, measurement of water quality is in terms of mass per unit volume; for example, the concentration of total suspended solids (TSS) or total nitrogen (TN) is measured in milligrams per liter (mg/L). The difference between inflow and outflow is then attributed to removal by the wetland. Instead, a mass balance, or budget, for each constituent is preferable (Kadlec and Knight 1996). A mass balance of a given nutrient in a wetland includes measurements of inputs via hydrologic pathways and outputs via hydrologic and atmospheric pathways. Measurement of cumulative flux into storage compartments (soils, vegetation, and plant litter) is desirable; however, rates of flux and turnover times are difficult to measure in situ (but see Fisher and Reddy (2001) for a description of phosphorus flux from wetland soils in the Everglades). Instead, measurements of standing stocks are more common, giving a snapshot of the retention of nutrients or sediments (Johnston 1991).

In order to compare removal efficiencies of wetlands, nitrogen or phosphorus inputs and outputs should be measured in terms of mass per unit area of wetland per year. Although most studies of wetland water quality measure removal efficiencies based on concentrations, recent work has focused on mass balances per unit area of wetland per year (e.g., Craft and Casey 2000, White et al. 2000; see Mitsch et al. 2000, Saunders and Kalff 2001 for reviews). Mass balances are often calculated for the growing season only, ignoring fall and winter inputs and outputs and leading to incomplete mass balances. Hydrology has a direct influence on the retention or export of nutrients and sediments (e.g., Devito and Dillon 1993a); thus, it is necessary to first understand a wetland's water mass balance before calculating nutrient mass balances (Kadlec and Knight 1996).

### *1. Nutrient Assimilation*

Wetlands are extremely complex in their ability to assimilate nutrients depending on their position in the landscape, watershed hydrology, groundwater flow path, and sediment type, location and permeability (Hill 1996, Devito et al. 2000, Hill 2000). Similar wetlands may exhibit different biogeochemical behaviour because of how they are linked to their watersheds (Hill and Devito 1997; Bedford 1999). Several characteristics contribute to wetlands' roles as nutrient sinks. In general, they accumulate organic matter, retaining nutrients in buried sediments; they are usually isolated from high-energy hydrodynamics (waves, currents, etc.) so promote sedimentation of organic matter; and their shallow water depth maximizes water-soil contact and therefore microbial processing of litter (Mitsch et al.1989). Other factors that influence nutrient assimilation by wetlands include: nutrient loading rate, water quality and depth, soil type and chemistry, vegetation, algal and microbial communities, primary production and decomposition rates, and hydraulic retention time (Moustafa 2000). A detailed examination of all of these factors will not be attempted here, but several examples of research conducted in these areas can be found in the primary scientific literature and are cited below.

Seasonal patterns of nutrient uptake and release further contribute to a wetland's ability to improve water quality. During the growing season, uptake and immobilization by microflora (bacteria and algae) and macrophytes retain nutrients; the dieback of plants in the fall releases nutrients to the water column through decomposition when they cannot be used for primary productivity (Mitsch et al. 1989). Conversely, uptake by plants and other aquatic organisms results in the conversion of inorganic nutrients to organic forms which can result in a net export of nutrients from a wetland during certain seasons (Devito and Dillon 1993a; Devito and Dillon 1993b; Devito et al. 1989).

Mitsch and Gosselink (2000a) reviewed a number of studies that estimated the area of wetlands required in a watershed to improve nutrient retention (nitrogen and phosphorus) and general water quality. Several examples from Midwestern USA and Scandinavia suggest that a range of 3-7% (average approximately 5%) of temperate-zone watershed should be in wetlands to provide adequate water quality values for the landscape.

*Nitrogen*

Nitrogen is the focus of water quality concerns where large amounts of fertilizers are used on high input crops (MacDonald 2000). In a potato growing region near Alliston, ON, Hill (1982) reported nitrate contamination >10 mg/l (the maximum allowable concentration in drinking water in Ontario (OME 2000)) of a shallow water-table aquifer underlying a sand plain and suggested that fertilizers are the major source of nitrate ($NO_3$-) contamination. On the prairies of North America, up to 50% of the nitrogen in fertilizers applied to crops may be lost in runoff, primarily in the form of nitrate (Neely and Baker 1989). Excess nitrate in runoff can then enter surface waters, contributing to eutrophication, or leach into groundwater where it may contaminate drinking water sources.

In agricultural areas without excess water such as Saskatchewan, water contamination by nitrogen under current management practices is associated with specific events such as storms and in areas with intensive livestock or crop operations (MacDonald 2000). In a survey of drinking water wells in Alberta, 13% of 376 shallow wells sampled had nitrate-plus-nitrite levels above the guideline for human drinking. Thus, prairie groundwater resources are not only at risk, but are already showing signs of nitrate contamination.

High levels of nitrate in drinking water can be toxic to humans causing methylglobanemia, or blue baby syndrome, wherein the oxygen carrying capacity of hemoglobin is blocked, causing suffocation (Naiman et al. 1995; Environment Canada 2001). Seventeen percent of Ontario farmland is at high risk for nitrogen contamination of waterways, particularly in southwestern Ontario, the Lake Simcoe region, and the South Nation watershed (MacDonald 2000). In a survey of drinking water wells in Ontario townships where over 50% of land area was under agricultural production, Goss et al. (1998) found that 14% of farm wells contained nitrate levels greater than the maximum allowable concentration in drinking water.

Nitrogen (N) in wetland soils and biota is primarily organic (Kadlec and Knight 1996). A wetland N cycle typically consists of interconversion between organic N, ammonium, and nitrate (Figure 4). The form of nitrogen most readily available for uptake by wetland microorganisms and plants is ammonium ($NH_4$+), produced by microbes during organic matter decomposition. Ammonium is absorbed by plants or microorganisms and the nitrogen incorporated into organic matter. As a positively-charged ion, ammonium can also be immobilized onto negative soil particles (Mitsch and Gosselink 2000b). In the soil, ammonium diffuses upward to the thin oxidized layer at the sediment-water interface. There, ammonium is oxidized through the process of nitrification to nitrite ($NO_2$-), then to nitrate ($NO_3$-). Nitrate must be reduced to ammonium by plants or microbes before it can be used in their growth (Mitsch and Gosselink 2000b).

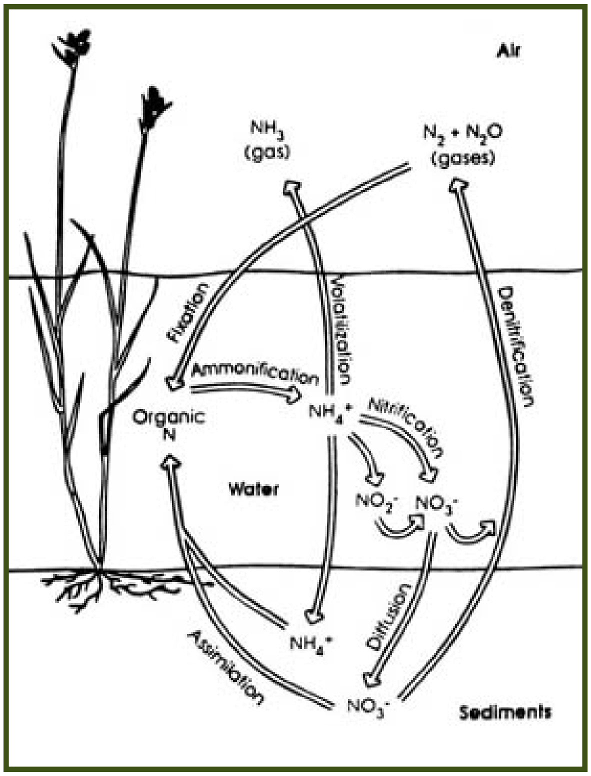


**FIGURE 4**

**Simplified wetland nitrogen cycle.** *(Knight and Kadlec 1996)*

Nitrogen is assimilated by wetlands primarily by one of three processes: (i) denitrification of nitrate to nitrogen gas ($N_2$) that is lost to the atmosphere, (ii) sedimentation of particulate N, or (iii) assimilation by plants and microbes (Saunders and Kalff 2001, Braskerud 2002, Janzen et al. 2003). Although N is retained in wetland plants, biota, and sediments, it may be permanently removed by denitrification and harvest of wetland plants (Matheson et al. 2002). Denitrification is typically the primary mechanism of N attenuation in wetlands and riparian buffers (see Section V), and is especially efficient for reducing nitrate contamination of shallow groundwater (Vellidis et al. 2003, Matheson et al. 2002, Flite III et al. 2001, Hanson et al. 1994).

Prairie wetlands are rarely unimpacted systems and can be expected to receive significant external nitrate loading from surrounding agricultural lands (Crumpton and Goldsborough 1998). However, there is evidence that wetlands are effective nitrate sinks in agricultural landscapes (Crumpton and Goldsborough 1998; Mitsch and Gosselink 2000b, 707). Crumpton and Goldsborough (1998) reviewed several studies of prairie potholes receiving sustained nitrate loads, and found that upwards of 80% of nitrate loading could be lost through denitrification.

The effectiveness of prairie wetlands as sinks for non-point source nitrogen loads is likely to depend on the magnitude of nitrate loads and the capacity of the wetlands to remove nitrate by dissimilatory processes (Crumpton and Goldsborough 1998). Increased nitrate loading in agricultural watersheds can be expected to stimulate denitrification (Isenhart 1992; Moraghan 1993). Recent studies suggest that the ability of natural, restored, and constructed wetlands to attenuate N loading from wastewater and surface water runoff from various land uses is high (Tables 2, 3), although there is some variability due to seasonal and landscape influences (e.g., White and Bayley 2001).

Cey et al. (1999) studied groundwater flow and geochemistry in the riparian/wetland zone of a small agricultural watershed near London. They found that increased recharge at the riparian/wetland zone, as compared to the artificially drained field, caused nitrate-rich groundwater from the adjacent field to be diverted downward beneath the wetland where it was attenuated by denitrification in the downward moving groundwater.

Mitsch et al. (2000) reviewed the nitrogen retention of wetlands (primarily constructed wetlands) and concluded that nitrate retention was clearly temperature (season) dependent. In the cold climate of the eastern USA, nitrate retention rates in constructed wetlands are on the order of 10 to 40 g-nitrogen/m2/yr and are sustainable for the treatment of non-point source (NPS) pollution. Saunders and Kalff (2001) reviewed several North American and European nitrogen mass balance studies, and found that, on average, N retention in wetlands was 64% of TN loading, 34% in lakes, and 2% in rivers.

*Phosphorus*

Phosphorus (P) enrichment of surface waters, whether by agricultural runoff or by wastewater effluent, and the resultant increase in primary production may lead to many undesirable effects on aquatic systems. These include blooms of nuisance algae that clog water intakes, increased turbidity of water bodies, decline of aquatic macrophytes due to shading, and many other water quality concerns. Phosphorus retention is considered one of the most important attributes of natural and constructed wetlands (Mitsch and Gosselink 2000b), and is key to determining downstream water quality (Reddy et al. 1999).

The primary forms of phosphorus that are biologically available for uptake by wetland plants and microorganisms are soluble inorganics (i.e., orthophosphates) (Mitsch and Gosselink 2000b). Total P is the sum of phosphorus dissolved in the water plus particulate phosphorus, including organic phosphorus, algal and bacterial phosphorus, and phosphorus sorbed to suspended solids (Kadlec and Knight 1996). All forms of organic P and insoluble inorganic P must first be transformed to ortho-phosphorus before they can be used by primary producers (Mitsch and Gosselink 2000b).

Phosphorus retention in wetlands is accomplished by three mechanisms: (1) adsorption onto peat and clay particles; (2) precipitation of insoluble phosphates with metals (iron, calcium and aluminum) under aerobic conditions; and, (3) incorporation into living biomass of bacteria, algae, and macrophytes (Mitsch and Gosselink 2000b, 186). The clay-phosphorus complex is particularly important because much of the phosphorus brought into wetlands is sorbed to clay particles. Phosphorus retention over the long term has been shown to be greater in floodplain wetlands than depressional wetlands in Georgia, due to the co-deposition of P with fine-textured clays in the floodplain wetland (Craft and Casey 2000). The primary means of net long-term storage of phosphorus is through wetland soil/sediment accretion (Kadlec and Knight 1996). For example, over 60% of P inputs from a beef processing facility were found to be stored in the sediments of a restored northern prairie marsh over a 5-year period (White et al. 2000). Most wetland macrophytes obtain phosphorus from soil; therefore, sedimentation of phosphorus sorbed onto clay particles is an indirect way in which phosphorus is made available to biotic components of the wetland (Mitsch and Gosselink 2000b). Plants transform inorganic phosphorus to organic forms that are stored in organic peat, mineralized by microbial activity, or exported from the wetland.

Johnston (1991) reviewed the retention of phosphorus of several wetlands in the US with no direct anthropogenic inputs and found that percent retention ranged from 9 to 80%. A similar range of percent retention of P can be found in the recent literature (Table 3). Schaefer et al. (1996) quantified the role of wetlands in buffering rural NPS phosphorus in the Eramosa River watershed in southern Ontario. They estimated that wetlands remove 92% of the phosphorus received directly from overland runoff, translating to a 46% reduction in potential phosphorus loads to the Eramosa River.

Phosphorus retention by wetlands in Ontario is variable, depending on season, stream flow, and other variables. The Hidden Valley wetland in Kitchener retained total phosphorus (inputs exceeded outputs by 100%), but exported plant-available ortho-phosphorus (Gehrels and Mulamoottil 1989). The majority of this export occurred in the fall, suggesting that potential eutrophication downstream of the wetland would be negligible because the growing season had ended due to low water temperature. In contrast, beaver ponds and conifer swamps in central Ontario's Precambrian Shield retained plant-available soluble phosphorus during summer, but overall retention of total phosphorus was low on an annual basis (Devito and Dillon 1993a, 1993b; Devito et al. 1989).

**TABLE 3 – Percent reduction in total nitrogen (TN), nitrate ($NO_3$), ammonia ($NH_4$), total phosphorus (TP), phosphate (SRP), sediment (TSS) and pathogens in natural wetlands.**

| Author | Location | TN | $NO_3$ | $NH_4$ | TP | SRP | TSS | Pathogens |
|---|---|---|---|---|---|---|---|---|
| Casey and Klaine (2001) | South Carolina; riparian wetland receiving golf course runoff | | 80 | | | 74 | | |
| Comin et al. (2001) | Spain; restored wetland receiving rice field runoff | 50-98 | | | | <50 | | |
| Jordan et al. (2003) | Maryland; restored wetland in agricultural watershed (two year average) | | 35 | 25 | | | 0 | |
| Kao and Wu (2001) | North Carolina; natural wetland receiving stormwater runoff from agricultural land | >80 | | | 59 | | 91 | |
| Nõges and Järvet (2002) | Estonia; natural riparian wetland receiving municipal wastewater (mass balances) | 65 | | | 17 | ++ | 96 | 99[a] |
| Shan et al. (2002) | China; natural multi-depression wetland system receiving continuous surface runoff | | | | 93.9 | 90.0 | 94.9 | |
| Velledis et al. (2003) | Georgia; restored riparian wetland adjacent to manure application area | | 78 | 52 | 66 | 66 | | |
| White and Bayley (2001) | Alberta; restored marsh receiving wastewater; summer | | 87 | 76 | 64 | | | |
| White and Bayley (2001) | Alberta; restored marsh receiving wastewater; winter | | -26 | 46 | | 26 | | |
| Woltemade (2000) | Maryland; restored wetland receiving agricultural runoff | | 68 | | 43 | | | |
| Woltemade (2000) | Illinois; restored wetland receiving agricultural runoff | | 36-45 | | 20 | | | |

++ *soluble phosphorus increased*  **a** *coliform bacteria*

Mitsch et al. (1989) studied the Old Woman Creek wetland in Erie County, Ohio with respect to phosphorus retention. Nutrient levels in runoff entering the Old Woman Creek wetland are high; phosphorus loading is estimated to be 12 - 23 g-phosphorus/m$^2$/ yr. Ortho-phosphorus concentrations in the stream entering the wetland was found to be significantly greater than that leaving the wetland. Because flow data and total phosphorus were not measured, the net retention of phosphorus could not be calculated, but was estimated to be 5-7 g-phosphorus/m$^2$/yr, or 30-39%. If their estimation was correct, and if other Lake Erie wetlands retain phosphorus similarly to the Old Woman Creek, the authors concluded that the existing wetlands on the lake could be retaining 75-100 tons/yr, or about 3.5 - 5% of the total NPS loading of phosphorus to the lake. Restoration of one-fourth of the original wetland area could possibly lead to a 24 - 33% reduction in phosphorus loading to western Lake Erie (Mitsch et al. 1989); other estimates of phosphorus retention by wetlands in the Laurentian Great Lakes suggest that 15% of watershed area should be maintained as wetlands (Wang and Mitsch 1998). Reeder (1994), in a study on the same wetland, found that phytoplankton productivity could account for gross uptake of up to 15 g-phosphorus/$m^2$/ yr. Macrophytes, which have traditionally been cited as critical components of maintaining water quality, accounted for only 0.1 g-phosphorus/$m^2$/yr. Reeder concluded that wetlands dominated by deep water and phytoplankton may be efficient traps for phosphorus in runoff.

In a review by Mitsch et al. (2000) that focused on the nitrogen and phosphorus retention of wetlands (primarily constructed wetlands), they concluded that phosphorus retention was highly variable from site to site (ranged from 0.4 to 47 g-phosphorus/$m^2$/yr) depending on soil chemistry, ambient water quality and water column productivity. Sustainable phosphorus retention, at least in constructed wetlands, appears to be in the range of 0.5 to 5 g-phosphorus/$m^2$/yr.

***2. Sediments***

Sedimentation is a major water quality concern in Canada and the U.S. In fact, excessive sediment loading from eroding land is considered the major pollutant of wetlands, lakes, rivers, and estuaries in the U.S. (Gleason and Euliss 1998). Of ten states reporting causes of wetland degradation to the United States Environmental Protection Agency (U.S.E.P.A.), nine states cited sedimentation or siltation as the most widespread cause of degradation followed by filling/draining and flow alterations (U.S.E.P.A. 2000).

Sediment consists of particles of all sizes, from fine clay particles to silt, sand, and gravel. Sedimentation and siltation of these particles and organic matter can cause damage to aquatic ecosystems, including clogged fish gills, suffocation of bottom-dwelling (benthic) organisms, reduction in fish reproductive habitat (benthic substrata), reduced water clarity, reduced primary productivity due to physical burial and reduced light availability, transport of chemicals attached to sediment particles, and the gradual infilling of water bodies (Gleason and Euliss 1998; U.S.E.P.A. 2000; Meyer et al. 2003). Water bodies located in agricultural landscapes are prone to receiving high sediment loads due to alteration of wetland catchment areas and cultivation of grasslands that once protected soils from erosion (Gleason and Euliss 1998).

Hydrology is a primary determinant of the sediment-retention capacity of a wetland (Brown 1988, Johnston 1991). Hydrology controls the source, amount and spatial and temporal distribution of sediment inputs to wetlands and other receiving water bodies (Johnston 1991). As water flows into a wetland, vegetation disperses the water and reduces flow velocity, and therefore increases the retention time of water in the wetland (Winter and Woo 1990). Reduced water velocity and increased retention time have a positive effect on sedimentation rates (Brown 1988; Hammer 1993), and in turn on the removal of sediment-associated pollutants such as nitrogen, phosphorus, pathogens, and pesticides. Particle size and soil properties of the surrounding watershed also influence sedimentation rates (Boto and Patrick 1978). Re-suspension of sediment will depend on the hydrological characteristics of the wetland, wetland size, area of open water, and wind and wave action.
Natural and constructed wetland systems are effective for sediment removal, typically measured by percent retention of total suspended solids (Tables 2, 3). Sediment retention can range between 49 and 98% in surface-flow and subsurface-flow constructed wastewater wetlands (Mitsch and Gosselink 2000b). Kadlec and Knight (1996, 331) found that reduction of suspended solids in wastewater and stormwater ponds ranged from 66-92%. Depressional wetlands (i.e., closed basin with no outlet) may retain all incoming sediment (Novitzki 1979; Gleason and Euliss 1998). Slope wetlands also retain sediment if water velocities decrease substantially within the wetland area (Novitzki 1979). Small riparian wetlands are also known to act as net sediment storage sites (Heimann and Roell 2000).

In Wisconsin, watersheds containing 40% wetland and lakes had sediment loads 90% lower than watersheds with no wetlands or lakes; only 5% of the wetlands were found to be responsible for trapping up to 70% of the sediment (Novitzki 1979). Novitzki (1979) determined that sediment retention could be maximized by maintaining a 10% cover of wetlands within a watershed. Other researchers have shown that the position of wetlands in the watershed can be more important than the extent of wetland area in terms of reducing sediment and nutrient loads; i.e., downstream wetlands have a greater effect on water quality (Johnston et al. 1990).

The ability of wetlands to remove and retain sediments is a basic concept of improved water quality, but excessive sediment loads can be harmful to natural wetlands. Many prairie wetlands are closed systems that can totally fill with sediments and hence lose their capacity to function properly (Gleason and Euliss 1998). Wetlands in agricultural watersheds in the Great Lakes region exhibit high turbidity, suspended solids, and nutrient levels (Crosbie and Chow-Fraser 1999). The trade-off between the importance of sediment removal as a water quality benefit and maintaining the topographic life of wetland basins needs to be integrated into management strategies of wetlands and watersheds (Gleason and Euliss 1998).

***3. Pathogens***

Many infectious diseases are transmitted through animal and human feces. Waterborne pathogens of serious risk to humans include strains of bacteria such as *Escherichia coli*, *Salmonella typhi*, *Campylobacter* species, and others; viruses such as enteroviruses, Hepatitis A, and others; and the protozoans *Entamoeba histolytica*, *Giardia intestinalis*, and *Cryptosporidium parvum* (Kadlec and Knight 1996; WHO 2000). These pathogens are persistent in water supplies due to their ability to survive outside of host organisms. Fecal contamination of natural surface and groundwater can be a serious problem in agricultural landscapes dominated by livestock production, and in highly populated areas where secondarily-treated wastewater characterized by abundant pathogens is often discharged directly to rivers, streams, or lakes. For example, in southern Ontario, Goss et al. (1998) found that over 34% of domestic wells in agriculturally-dominated landscapes contained levels of coliform bacteria greater than the maximum allowable concentration in Ontario drinking water (OME 2000). Natural bacteria populations are generally low in wetlands but they may be variable and seasonally high in certain wetlands because of wildlife populations (e.g., staging waterfowl) (Kadlec and Knight 1996).

The ability of constructed wetlands to reduce populations of pathogenic microorganisms in wastewater effluent has been demonstrated globally (e.g., Kadlec and Knight 1996; Schreijer et al. 1997; Stott et al. 1997; Hill and Sobsey 1998; Decamp and Warren 2000; Neralla and Weaver 2000). Many of the processes that reduce pathogen populations in natural systems are equally or more effective in wetland treatment systems (Kadlec and Knight 1996). Factors influencing removal of pathogens include: natural die-off, sedimentation, predation, and adsorption, which are in turn influenced by retention time and seasonal variability (Falabi et al. 2002). Macrophytes are essential because they provide surface contact area for microbes that mediate most of the nutrient and pollutant transformations that occur in wetlands (Hamilton et al. 1993). Vegetated wetlands appear to be more effective for pathogen removal than facultative ponds and other natural treatment systems that have less physical contact between pathogens and solid surfaces (Kadlec and Knight 1996).
Treatment wetland removal efficiencies are nearly always greater than 90% for coliforms and greater than 80% for fecal streptococcus (Kadlec and Knight 1996). *Giardia, Cryptosporidium* and *Salmonella* are also reduced effectively by wetlands (Table 2). Few studies of pathogen removal by natural wetlands are found in the literature, thus additional information is necessary to confirm that natural wetlands are as effective as constructed wetlands.

***4. Contaminants***

The ability of wetlands to degrade and remove contaminants such as pesticides, metals, landfill leachate, and urban stormwater runoff has been examined in natural wetlands (e.g., Fernandes et al. 1996, Goldsborough and Crumpton 1998), and to a much greater extent in constructed wetlands (e.g. Hammer 1989, Kadlec and Knight 1996). Pesticides are chemicals that are toxic to living organisms, and are targeted at either plants (herbicides), fungi (fungicides), or insects (insecticides) (Goldsborough and Crumpton 1998). Landfill leachate and urban stormwater runoff often include mixtures of toxic substances including metals, household chemicals, hydrocarbons, salt, and sand. Wetlands have been shown to attenuate landfill leachate near Pembroke ON (Fernandes et al. 1996), dissolved chlorinated volatile organic compounds in groundwater near a former manufacturing site in Minnesota (Richard and Connell 2001), and heavy metals from urban stormwater runoff and a former lead-acid manufacturing plant (Sriyaraj and Shutes 2001, Gallardo-Williams et al. 2002).

Transport of pesticides into water bodies occurs by direct overspray, by aerial drift of pesticide droplets, by wind drift of particulates to which pesticides are adsorbed, by dissolution in surface water runoff, snowmelt, or groundwater (Waiser and Robarts 1997; Goldsborough and Crumpton 1998), or by accidental spills. Various studies of pesticide residues in wetlands of the Great Plains have reported moderate to high frequencies of detection, up to 100% in the case of the herbicide 2,4-D in Saskatchewan farm ponds (Grover et al. 1997). Although Nebraska wetlands surrounded by cropland had significantly greater atrazine concentrations, 94% of the sampled wetlands contained detectable levels of herbicides, regardless of surrounding land use(Frankforter 1995; also see Donald et al. 2001). Frank et al. (1990) compiled results of pesticide surveys conducted in rural ponds in Ontario between 1971 and 1985. Landowners contacted the Ministry of Agriculture or Environment when they suspected a pond had been contaminated by pesticides. Of the 211 ponds sampled, 132 or 63% were contaminated by at least one pesticide. Goldsborough and Crumpton (1998) argue that wetlands have specific characteristics that increase pesticide dissipation through photolysis and adsorption as compared to other water bodies. The high levels of biological productivity in wetlands results in profuse submersed and emergent plant growth. This increases the availability of surface area for adsorption, plant sequestration, microbial degradation, and exposure to light. Many studies have shown the ability of submersed macrophytes to remove pesticides and thus prevent further negative effects on aquatic biota (e.g., Brock et al. 1992; Karen et al. 1998). Highly organic wetland sediments also are preferential adsorption sites for pesticides (e.g. Brock et al. 1992). The shallow nature of wetlands increases light penetration, and thus increases the potential for photolysis. Wetlands in agricultural landscapes have high potential for intercepting and dissipating pesticides. For example, Kao et al. (2001, 2002) found that a natural wetland in North Carolina completely removed atrazine from diffuse agricultural runoff after several storm events.

### *d) Summary*

The hydrological functions of wetlands include storage and eventual release of surface water, recharge of local and regional groundwater supplies, reduction in peak floodwater flows, de-synchronization of flood peaks, and erosion prevention. Many wetlands are known to provide any or all of these functions; each situation is uniquely dependent on local topography, climate, geology, and watershed characteristics. Position in the landscape, location of the water table, soil permeability, slope, and moisture conditions influence the ability of any given wetland to attenuate floodwaters. Wetlands commonly retain part of surface inflow and release the water during an extended period resulting in a peak flow lag behind the initial peak runoff into the wetland. As surface water enters a wetland, the vegetation can disperse the incoming water, reduces the flow velocity, and thus increases residence time of water in the wetland. Water storage in wetlands is underground or in surface depressions and when the water table is low considerable storage capacity is available. Wetlands that are saturated may have little capacity to store water. Wetland channelization reduces the ability of a wetland to attenuate runoff during flood conditions. Maintaining and restoring wetlands on the landscape reduces river flow rates and flooding.

Recharge of groundwater is an extremely important function of some wetlands; water percolates slowly from wetlands to aquifers. Movement of groundwater is related to soil permeability and local topography. Groundwater recharge occurs from many areas in the landscape, including wetlands (from seasonal to permanent), uplands, and areas of extreme permeability such as sand deposits. Interactions between wetlands and local or regional groundwater supplies are complex and site-specific. Some wetlands receive significant groundwater discharge. The interactions of wetlands and groundwater are affected by the position of the wetland with respect to groundwater flow systems, geologic characteristics of the substrate and climate.

Wetlands are extremely complex systems and several characteristics contribute to their roles as nutrient sinks. They accumulate organic matter, retain nutrients in buried sediments, convert inorganic nutrients to organic biomass, promote sedimentation of solids, and their shallow water depth maximizes water-soil contact and therefore microbial processing of nutrients and other material in the overlying water. Wetlands can be effective nitrogen sinks in agricultural landscapes (Table 4) due to assimilation by microbes and denitrification. Other wetlands may retain nitrate and ammonium but may export organic nitrogen. Phosphorus retention in wetlands is accomplished through adsorption onto organic peat and clay particles, precipitation of insoluble phosphates with metals and incorporation into living biomass. Phosphorus retention rates for wetlands can be significant (Table 4). Wetlands are hydrologically, chemically and biologically linked to the landscape in which they occur and have variable nutrient-retention efficiencies depending on their position in the landscape, watershed hydrology, hydrogeologic characteristics and climate.

**TABLE 4 – Range of percent retention for nitrogen, phosphorus, sediment, coliforms and pesticides in natural wetlands.**

| | | Retention (%) |
|---|---|---|
| Nitrogen | – Nitrate | up to 87 |
| | – Ammonium | up to 76 |
| Phosphorus | | up to 94 |
| Sediment | | up to 98 |
| Coliforms (constructed wetlands) | | up to 99 |
| Pesticides | | <1 day - several months[1] |

**1** *Time for residues to decrease by 50%*

Wetlands can reduce the impacts of sedimentation on water quality within watersheds (Table 4). Hydrology is a primary determinant of the sediment-retention capacity of a wetland and controls the source, amount, and spatial and temporal distribution of sediment inputs. Wetland vegetation is important because it disperses the water and reduces flow velocity that increases the retention time of the water in the wetland, resulting in increased sediment deposition. Percent of wetland area and position in the landscape are important for reducing sediment loads.
Little information exists on the effects of the ability of natural wetlands to reduce microbial populations in water. The effectiveness of constructed wetlands to reduce pathogenic organisms from wastewater is high (Table 4). Natural wetlands are dominated by microbes (bacteria, fungi and algae) and plant life that are important for reducing pathogens.

High levels of biological productivity in wetlands result in dissipation of pesticides due to profuse submersed and emergent plant growth that increases the availability of surface area for pesticide adsorption, plant sequestration, microbial degradation, and exposure to light. In general, common pesticides of surface and groundwater disappear rapidly from wetlands (Table 4), primarily due to adsorption to organic matter in sediments and decomposing litter.

# V riparian area management

Riparian areas are transitional landscape features occurring between uplands and wetlands, streams, or lakes; it is this position in the landscape that allows them to act as natural “filters” of both surface and groundwater. Riparian zones are typically characterized by soils vegetation and biota that are considered transitional between upland and wetted habitats. Natural riparian areas have been altered by activities that have modified the landscape, including industry, agriculture, and urban development; however, restoration and conservation of remaining riparian zones have accelerated as our understanding of their critical role in watershed functioning expands.

Buffers are areas of native or replanted perennial vegetation that lie between lands subject to human alteration and naturally occurring waterways, and may be referred to as buffer strips, riparian buffers, or grass/vegetated filter strips (VFS) (Castelle et al. 1994; Dosskey et al. 2002). Buffers are critical for abatement of non-point source (NPS) pollutants in both surface and groundwater; in fact, the USDA has developed two national standards, in the form of filter strips and riparian forest buffers, toward reducing agricultural NPS pollution (Lee et al. 2003). Because buffers typically are components of agricultural BMPs, agriculture will be the focus of the following discussion; however, other industries and land use classes also find applications in which natural and restored riparian buffers are useful for NPS pollution prevention in nearby waterways.

Buffers reduce surface water runoff, thereby increasing sedimentation and retention of sediment-associated pollutants (nutrients, pesticides, bacteria, etc.). Buffer strips physically act as holding areas, where the presence of vegetation reduces surface runoff by improving infiltration, enhancing evapotranspiration, and intercepting rainwater (Flannagan et al. 1989; Munoz-Carpena et al. 1993; Mendez et al. 1999). A decrease in runoff volume and velocity as water moves through the buffer allows for sediment and associated pollutants to deposit in the buffer and increases the time of contact for adsorption onto soil and vegetation (Fajardo et al. 2001; Rankinen et al. 2001). This results in a reduction in surface runoff and associated pollutants to down-slope riparian systems (Hayes et al. 1979; Foster 1982; Rankinen et al. 2001). Retention of sediment by buffers in literature reviewed here typically is high, whereas percent retention of nitrogen, phosphorus, pesticides, and fecal coliform bacteria is variable (Table 5).

Because nitrate is primarily exported from watersheds via groundwater, the ability of riparian areas to reduce nitrate concentrations has been of great interest. Although the species composition of riparian vegetation community is important, nitrate removal capacity is dependent on the interaction of groundwater with “biologically active zones” - riparian zone components that support removal processes such as plant/microbial uptake and denitrification (Gold et al. 2001). Site attributes such as hydric status and geomorphology affect this interaction (Rosenblatt et al. 2001), and therefore should be incorporated into efforts to integrate riparian zones into watershed scale nitrate management schemes.

In addition to their importance in water quality, riparian buffers also have a cooling effect on the water temperatures in adjacent riparian zones (such as streams), the result of shading of surface water runoff as it moves over land. This has been shown to have a beneficial impact on the population of certain fish species in Ontario (Barton et al. 1985).

By combining the needs of various wildlife species, the goals for nutrient retention and the land availability, buffer strips could be effectively integrated in the landscape (Fennessy and Cronk 1997). The size of the buffer required is determined by a number of factors: the type of vegetation present, the extent and impact of the adjacent land use, and the functional value of the receiving wetland. Variations in these factors will affect each buffer's capacity to improve surface water quality as water moves through the buffer. Since the slope of a buffer strip is difficult to manipulate, altering the buffer width seems the most promising means to optimize effectiveness. An insufficiently small buffer may put an aquatic resource at risk where an excessively large one will unnecessarily pull land out of agricultural use (Castelle et al. 1994).

Buffers may be positioned in the landscape depending on local physiographic and hydrological features, ranging from within- and edge-of-field to streamside. Although the methods of determining appropriate, efficient, yet cost-effective buffer dimensions and biological components are beyond the scope of this document, there is still considerable research required to ease planning and application of buffers to reduce agricultural NPS pollution (Dosskey 2002). Lyons et al. (2000) reviewed the positive and negative implications of grassed, treed, or mixed riparian buffers. Site-specific studies of optimal buffer width and vegetation type are available (e.g., Dukes et al. 2002, Sparovek et al. 2002) as are discussions related to management and restoration of riparian buffers (Simpkins et al. 2002, Quinn et al. 2001, Hession et al. 2000, Jorgensen et al. 2000, Lowrance et al. 2000a).

**TABLE 5 – Percent reduction in groundwater and surface water total nitrogen (TN), nitrate-N, TKN, total phosphorus (TP), phosphate ($PO_4$) and sediment (TSS) in buffer strips.**

| Study | Parameter | Reduction (%) | Notes |
|---|---|---|---|
| ***Surface Water*** | | | |
| Ontario (Abu-Zreig et al. 2003) | TP | 31-89 | VFS vs. bare soil controls. |
| Iowa (Lee et al. 2003) | TN | 80, 94 | In grass and grass/woody buffers respectively. |
| | Nitrate-N | 62, 85 | |
| | TP | 78, 91 | |
| | $PO_4$ | 58, 80 | |
| | TSS | 95, 97 | |
| Norway (Syverson 2002) | TP | 76, 89 | In 5m and 10m buffer, respectively. Avg. 1992-99. |
| | TN | 62, 81 | |
| | TSS | 81, 91 | |
| | Organic M | 83, 90 | |
| Connecticut (Clausen et al. 2000) | Nitrate-N | 83 | Restored riparian buffer vs. row crop. |
| | TKN | 70 | |
| | TP | 73 | |
| | TSS | 92 | |
| ***Groundwater*** | | | |
| Estonia (Kuusemets et al. 2001) | TN | 40, 85 | In 31m and 51m buffers, respectively. |
| | TP | 78, 84 | |
| Neuse R. Basin, North Carolina (Spruill 2000) | Nitrate-N | 65-70 | Riparian buffers vs. non-buffer areas. |
| Connecticut (Clausen et al. 2000) | Nitrate-N | 35 | Restored riparian buffer vs. row crop |
| Virginia (Snyder et al. 1998) | Nitrate-N | 45 | Riparian buffer vs. upland agricultural field. |

The efficiency of riparian buffers determined in laboratory- and field- level experiments is not always demonstrated at the watershed scale, and may be partially due to spatial heterogeneity in hydrology and landforms (Montas et al. 2000). For example, Schiff et al. (2002) found that two adjacent forested catchments in Ontario had annual nitrate export that differed by a factor of ten, although soils, forest cover, and microbial nitrification were similar in each watershed. The difference was attributed to slope stratigraphy and hydraulic conductivity, which influenced groundwater flow in relation to the biologically active zone. Research at this scale is lacking, as most of the quantitative studies of the ability of buffers to abate water pollution focus on within-field processes, instead of examining the response of streams or lakes to buffer placement (Dosskey 2001).

Mathematical models are an alternative way to develop estimates of water quality improvement in response to buffers. The Riparian Ecosystem Management Model (REMM) was designed to simulate biological, chemical, and physical processes that occur in riparian buffer zones, and allows for comparisons of management scenarios and the incorporation of site-specific conditions into buffer zone design (Lowrance et al. 2000a). When used along with pollutant loading models such as GLEAMS (Groundwater Loading Effects of Agricultural Management Systems), REMM may be used to estimate nutrient loading from agricultural fields through riparian buffer zones (e.g., Stone et al. 2001).

### *a) Sediment Removal and Erosion Control*

Buffers control erosion by blocking the flow of sediment and debris, by stabilizing wetland edges and stream banks, and by promoting infiltration (Shisler et al. 1987). Bharati et al. (2002) found that cumulative infiltration of surface runoff was 5x greater under riparian buffers than within cultivated fields or pastures in Iowa. Buffers form a physical barrier that slows surface flow rates and mechanically traps sediment and debris. Roots maintain soil structure and physically retain erodible soil. Wilson (1967) concluded that buffer width, sediment load, flow rate, slope, grass height, and density all affect sediment removal. Simulated VFS experiments in laboratory flumes also suggest that density, slope, and sediment particle size are major factors determining sediment deposition in buffers (Jin and Romkens 2001). Sediments and NPS pollutants are trapped by buffers most efficiently when field runoff is dispersed uniformly (i.e., when concentrated flows do not occur) (Dosskey et al. 2002).

The buffer width required for efficient nutrient/sediment removal has been debated (Fennessey and Cronk 1997). Subsurface flows may be more effective than surface flows for nitrate removal, and removal increases as buffer width increases. Many studies have found the bulk of nitrate sediment removal occurs in the first few meters of the buffer zone (Dillaha et al. 1989; Peterjohn and Correll 1984; Ghaffarzadeh et al. 1992). Conditions for denitrification are particularly optimum at the receiving edge of a buffer because carbon (required as an energy source) is abundant and vegetative growth is often most dense at the edge of the strip where nitrate enters (Fennessey and Cronk 1997).

Ghaffarzadeh et al. (1992) studied the effectiveness of two, 9.1 m grass vegetated filter strips for sediment removal. They found that 85% of the sediments were removed with no difference in sediment removal in either of the 2 buffers beyond a distance of 3.1 meters. Neibling and Alberts (1979) found sediment discharge reduced by over 90% in a 5 m grass buffer. Clay transport was reduced by 83%. Ninety-one percent of the incoming sediment load was removed in the first 0.6 meters of the buffer strip. Magette et al. (1989) found a 66% reduction in sediment passing through a 4.6 m grass buffer. Tate et al. (2000) reported that a buffer area excluding livestock from irrigated pasture in the Sierra Nevadas of California significantly reduced TSS concentrations and loads compared to an unbuffered control. Other recent studies have demonstrated sediment removal between 81 and 97% in various buffer types and sizes (Table 5).

### b) Nutrient Assimilation

Johnes et al. (1996) estimate 95% of cattle wastes, 85% of pig wastes and 90% of poultry wastes are returned to the land. Of this, up to 17% of nitrogen and 3% of phosphorus are thought to reach drainage networks. These numbers reflect trends occurring in North America and in Europe, particularly in the Netherlands and the United Kingdom. Wherever there is intensive cropping and livestock production occurring great potential exists for nutrient loading of receiving watercourses (Heathwaite et al. 1998; Cey et al. 1999).

In Ontario, water and sediment quality for 22 wetlands in the Great Lakes basin was researched by Crosbie and Chow-Fraser (1999). Concentrations of phosphorus, nitrogen, and inorganic suspended solids increased predictably as agriculture became the dominant land use in the respective watersheds. Their research found that the use of forested buffer strips in agriculturally dominated watersheds led to measurable improvements in the water quality of downstream wetlands and streams. These findings were echoed by research in South Dakota by Rickerl et al. (2000). Four wetlands, two buffered by pasture grass and two not buffered from upland agriculture, were compared for water quality. Concentrations of nitrate and phosphorus were significantly less in the buffered wetlands. They also detected more storage of nitrogen and phosphorus in the plants of the two wetlands that were not buffered from the surrounding uplands.

The variety of vegetative cover in a buffer strip may determine its efficiency in intercepting nitrate, ammonia or phosphorus (Fennessey and Cronk 1997). Forested buffer strips are more efficient in removing nitrate than herbaceous buffer strips (Haycock and Pinay 1993, Correll 1991, Vought et al. 1991). The roots and root exudates of the trees put more organic carbon in the soil profile providing the primary source of carbon required for the denitrification of nitrate (Schipper et al. 1991). Grass buffers appear to be more effective than mixed grassed buffers (grass plus forest buffers) for removing total organic nitrogen plus ammonium and sediments from surface water (Gilliam et al. 1997). Phosphorus retention appears to be maximized when buffer strips contain both woody and herbaceous vegetation (Vought et al. 1994, Osborne and Kovacic 1993).

#### *1. Nitrogen*

The mechanisms for nitrate removal by buffer strips are complicated, but vegetation uptake in the roots and anaerobic microbial denitrification in the saturated zone of the soil are considered to be the main mechanisms (White et al. 1997; Hill et al. 2000). Significant denitrification of subsurface groundwater nitrate has been observed in many studies, but generally is limited by differences in soil saturation and organic carbon content of riparian soils (Shannon et al. 2000, Flite III et al. 2001). Localized denitrification may occur in deeper groundwater where there are available organic carbon supplies (e.g., in a deep riparian aquifer in Ontario; Hill et al. 2000). Riparian zone hydrology also plays a role in the degree of denitrification of nitrate (Angier et al. 2001). Wigington, Jr. et al. (2003) found that, although nitrate was reduced in shallow groundwater moving from commercial grass fields through the herbaceous riparian zone, the overall potential for denitrification was limited because very little runoff actually contacted the riparian zone. The majority of overland flow moved to streams via saturated swales. They concluded that, in poorly drained landscapes, nitrate loading to streams may be reduced more effectively by correct fertilizer application rates and timing. Similar results in an urban riparian zone in Japan were reported by Kinohira et al. (2001).

Relatively narrow buffers seem to be very effective in reducing the amount of nitrate as surface waters move through them. In Wisconsin, Madison (1992) found that 4.6 m and 9.1 m grass vegetated filter strips reduced ammonium and nitrate by approximately 90 and 96%, respectively. Mander et al. (1997) compared a wet meadow/grey alder buffer strip (11 m and 20 m, respectively) to a wet meadow/grey alder/grass buffer (12 m, 28 m, and 11 m, respectively) in Estonia. The grey alder/wet meadow strip removed 67% of the total nitrogen and the wet meadow/grey alder/ upland grass combination was capable of removing 96% of the nitrogen. Dillaha et al. (1989) reported that a 4.6 m and a 9.1 m grass filter strip in Virginia removed an average of 54 and 73% of nitrogen. Young et al. (1980) found that the average reduction in total nitrogen associated with solids from feedlot runoff was 84% over 2 years using a 41 m cropped buffer system in Minnesota. Recent studies indicate that 62-94% of total nitrogen and 62-85% of nitrate in surface runoff are retained by buffers (Table 5). Removal of nitrate from groundwater ranged from 35 to 70% (Table 5).

*2. Phosphorus*

Inputs of phosphorus are often essential for profitable crop and livestock production; however, its export in watershed runoff can accelerate the eutrophication of receiving waters (Sharpley et al. 2000, Environment Canada 2001). Efforts to reduce phosphorus losses from agricultural systems need to balance off farm phosphorus inputs in feed and fertilizer with outputs in harvested products (Sharpley et al. 2000). This minimizes soil phosphorus inputs in excess of crop requirements. This approach combined with other practices such as crop residue management, conservation tillage and buffer strips can further reduce phosphorus loss via surface runoff and erosion (Chambers et al. 2000; Uusi-Kamppa et al. 2000).

Uusi-Kamppa et al. (2000) determined that grassed buffer zones, with widths up to 16 m, effectively reduced total phosphorus in runoff from agricultural land in both long-term and short-term experiments in Norway, Finland and Sweden. Retention of total phosphorus in buffers varied from 27 to 97%. Most phosphorus remained in the upper layers of the buffer zones regardless of buffer width. They recommended wider buffer zones in areas with poor soil infiltration and higher soil erosion (heavy clay soils). In Estonia, Mander et al. (1997) found that the grey alder/wet meadow strip (11 m and 20 m, respectively) removed 81% of the phosphorus and a wet meadow/grey alder/grass (12 m, 28 m, and 11 m, respectively) combination was capable of removing 97% of the phosphorus. Dillaha et al. (1989) reported that a 4.6 m and a 9.1 m grass filter strip removed an average of 61 and 79% of phosphorus in Virginia. Madison et al. (1992) trapped 99.9% of phosphorus using a 9.1 m filter strip in Wisconsin. He found no improvement in the trapping efficiency of phosphorus by increasing the buffer strip beyond 9.1 m. Recent work (Table 5) suggests that up to 91% of phosphorus in surface runoff may be retained by buffers. Although Dosskey et al. (2001) suggest that groundwater phosphorus tends to increase through buffers, very few studies of groundwater phosphorus dynamics have been completed (but see Carlyle and Hill 2001 for phosphate dynamics in a forested floodplain connected to a large sand aquifer in Ontario).

Young et al. (1980) found that the average reduction in total phosphorus associated with solids from feedlot runoff was 83% over 2 years using a 41 m cropped buffer system in Minnesota. Other research has shown that a 1:1 ratio of grass vegetated filter strip size to waste production area (cumulative surface area of animal pens) could result in a 90 to 100% reduction in nutrients level in runoff to adjacent riparian systems (Bingham et al. 1980; Overcash et al.1981).

### c) Pathogens

Bacteria loss in runoff from freshly manured soil can be as high as 90% (Crane et al. 1983). Early research by Dickey and Vanderholm (1981) and Walker et al. (1990) suggested that buffer strips alone would not reduce bacterial levels to water quality guidelines. For example, although tall fescue VFS reduced coliform bacteria by up to 87% in runoff from livestock confinement areas, numbers remained high and were in excess of 1000 CFU/100 mL (Fajardo et al. 2001). Coyne et al. (1995) found that 9 m buffers trapped up to 74% of fecal coliforms shortly after rain events on soil fertilized with fresh poultry waste. However, they noted that this 74% reduction in fecal coliforms resulted in more than 200 fecal coliforms/100 ml, thereby exceeding the minimum drinking water standards of 0/100ml set in Ontario. Young et al. (1980) evaluated a cropped buffer system over 2 years and found a reduction of 69% for total coliforms and fecal coliforms and 70% for fecal streptococcus.

Entry et al. (2000a,b) studied 30 m mix of grass and forested buffer strips applied with swine wastewater in Georgia. Total and fecal coliform numbers in the wastewater pulse did not decline as runoff moved downslope. Vegetation type in the buffer strips usually did not affect survival of total and fecal coliform bacteria in the soil. However, they found that decreasing soil moisture and increasing soil temperature substantially decreased survival of total and fecal coliform bacteria at different soil depths (0-5, 5-15, 15-30 cm). Soil moisture (dry) and temperature (>28 C) will effectively decrease survival rates of pathogenic bacteria. They recommended that waste applications to agricultural lands be conducted during optimal periods of warm-dry weather when soils are dry and bacteria are less likely to be transported. They also suggest that the buffer strip vegetation should have high evapotranspiration rates to reduce soil moisture. Selecting the appropriate vegetation type and increasing the buffer strip width can improve the efficiency of buffer strips for reducing pathogens (Jim Entry, US Department of Agriculture, personal communication).

Techniques are currently being developed to reduce pathogens in animal wastewater before they reach buffer strips and when used along with vegetative buffers may effectively reduce the input of pathogens from animal confinement areas to water resources (Entry and Sojka 2000; Sojka and Entry 2000).

### *d) Pesticides*

Herbicides are the most frequently detected pesticides in surface waters. The amount of pesticides applied, their solubility, persistence, degree of soil adsorption and their location in the soil profile determines their concentration in the sediment and water (Fawcett et al. 1994). The amount of pesticide transfer in runoff water depends on the soil adsorption properties of the pesticide. Most herbicides have intermediate adsorption to the soil and are lost primarily with surface water runoff (Baker and Laflen 1983; Gaynor et al. 1995). Of the total amount lost, 60 to 90 % of common herbicides such as atrazine, alachlor, cyanazine and metolchlor are lost in this water phase (Fawcett et al. 1994).

Gril et al. (1997) and Patty et al. (1997) reviewed the findings from a study in France on the ability of 6, 12, and 18 m grassed buffer strips to reduce lindane, atrazine and its metabolites in surface water runoff. Averaged between the different sized buffer strips, lindane and atrazine were reduced 72 to 100% and 44 to 100%, respectively. Grass buffer strips (20.1 m) in Iowa retained 11 to 100% of the atrazine, 16 to 100% of metolachlor, and 8 to 100% of cyanazine (Arora et al. 1996). Ranges in these percentages were the result of rainfall duration and intensity. Herbicide retention was less during peak flows and increased as the runoff event progressed (i.e., at lower flow rates). Infiltration was the key process for retention of the moderately adsorbed herbicides. Benoit et al. (1999) found a rapid degradation of the herbicide isoproturon (ISP) in a 5 m grass buffer strips down-slope from cropland in France. They found the half-life for ISP was 72 days in the cultivated soil compared to 8 days in the buffer strip soil. In addition to the shorter half-life of ISP, a large proportion of the ISP residue in the buffer strip bound to the soil and was no longer available to loss through surface water flows.

Mersie et al. (1999) found that atrazine and metolachlor were reduced in switch grass filter strips by 52 and 59%, respectively. Bare soil "controls" retained 41-44% of these pesticides. In a Nebraska study by Schmitt et al. (1999), sediment-associated permethrin was reduced by 27-83% in vegetated filterstrips, whereas dissolved constituents such as atrazine, alachlor, and bromide were not noticeably attenuated.

## e) Summary

Sustaining both agriculture and the integrity of aquatic ecosystems requires the improvement of surface and groundwater quality while maintaining farm productivity. Vegetated buffer strips can effectively control erosion by forming a physical barrier that slows the surface flow of sediment and debris, by stabilizing wetland edges and stream banks, and by promoting infiltration. The required width of a buffer size is determined by the type of vegetation present; the extent and impact of the adjacent land use; and the functional value of the receiving wetland. Many studies have found the bulk of sediment removal occurs in the first few meters of the buffer zone; sediment removal can be significant (Table 6).

**TABLE 6 – Range of percent retention for sediment, nitrogen, phosphorus, pesticides and coliforms in buffer strips.**

| | Retention (%) |
|---|---|
| Sediment | 66-97 |
| Nitrogen | 35-96 |
| Phosphorus | 27-97 |
| Pesticides | 8-100 |
| Coliforms [1] | 70-74 |

**1** *fecal coliform*

Buffer strips can effectively remove nutrients from surface water flow. The main mechanisms of nitrate removal are by vegetation uptake in the roots and anaerobic microbial denitrification in the saturated zone of the soil. Relatively narrow buffers seem to be very effective in reducing nitrogen (Table 6). Phosphorus retention can be effective (Table 6) in buffer strips that contain both woody and herbaceous vegetation, grasses and cropped buffer systems. Buffer strips can trap a significant proportion of pathogens (Table 6); however, remaining levels often exceed minimum drinking water standards. Low soil moisture and high soil temperature substantially decrease survival of total and fecal coliform bacteria. The key process for pesticide retention in buffer strips is infiltration. Grass buffer strips can reduce pesticides significantly (Table 6).

Buffer strips are an essential practice in watershed protection; however, they should be viewed as a secondary best management practice. In-field management practices such as conservation tillage and upland conservation are important for pollution control because they prevent pollution at its source.

vi

# permanent cover

## a) Upland Conservation Programs

Sustaining agriculture and watershed ecosystems requires the improvement of surface and groundwater quality while still maintaining farm profitability and rural vitality (Rickerl et al. 2000). Two upland conservation programs that have demonstrated this mix are the Beneficial Management Practice of conservation tillage and the Conservation Reserve Program (CRP) of the United States.

Conservation tillage is an agricultural practice that leaves significant crop residue such as stems, stalks and leaves on the soil surface from harvest to the next planting. Conservation tillage systems include no-till, ridge-till, strip till, mulch-till and reduced-till with no-till being the most beneficial to the environment. These practices retain a minimum of 30% of the soil surface covered with crop residue (Fawcett 1987; Edwards et al. 1988; Gowda et al 2003) with no-till leaving at least 75% of the surface covered (Edwards et al. 1988). No-till only disturbs the soil in a narrow (3-8 cm) seedbed (Fawcett et al 1994), created by coulters or seed furrow openers, attached to a planter (Uri 2000).

According to Statistics Canada's Agriculture Census (2001), conservation tillage and no-till seeding increased from 24% and 7% in 1991 to 30% for each in 2001. These conservation practices have historically been more common to the prairies but in 2001, eastern Canada reported a significant increase (Statistics Canada 2001).

The permanent cover program, CRP, was initiated in the United States in 1985 with the intention of retiring highly erodible/marginal farmlands to permanent grass cover or other perennial plantings (Randall et al. 1997). The Program was established to reduce erosion, protect soil productivity, reduce sedimentation, improve water quality, and improve wildlife habitat. Through a series of annual payments, CRP was designed to help landowners and operators conserve and improve soil and water resources on their farms and ranches while still maintaining an economic return. The United States Department of Agriculture's Farm Service Agency (2003) state active CRP contracts for all program years (1986 to 2004) total 34,120,492 acres (13,808,073 hectares).

Canada recently developed a program similar to the CRP. Greencover Canada is a new five-year, $110 million initiative under the Agricultural Policy Framework that will provide technical advice and financial assistance to producers converting environmentally sensitive land to perennial cover. The goals are to protect land from wind and water erosion, improve water quality, enhance biodiversity and increase carbon sequestration in soil (Agriculture and Agri-food Canada 2003).

Studies show the adoption of conservation tillage and permanent cover programs can reduce soil erosion, decrease nitrogen and phosphorus fertilizer runoff, and lessen the impacts of pesticides compared to conventional tillage methods (Davie and Lant 1994; Randall et al. 1997; Intarapapong et al. 2002; Fawcett and Towery 2003).

## b) Sediment Removal and Erosion Control

No-till conservation tillage and permanent cover programs like CRP help reduce erosion by protecting soil from the erosive capabilities of wind, rain and runoff.

Conservation tillage is one of the most practical and economical ways of reducing soil erosion. Several studies show that soil erosion can decrease by as much as 90 to 98% as a result of conservation tillage practices (Baker and Laflen 1983; Fawcett 1994; Clausen et al. 1996). Seta et al. (1993) evaluated the effects of conventional, chisel-plow and no tillage on the quality of runoff water. They found no-till reduced total sediment loss by 98%, and chisel-plow reduced it by 78% compared to conventional tillage. Intarapapong et al. (2002) examined the economic and environmental impacts of no-till practices for cotton, soybeans and corn in the Mississippi Delta. They found no-till cotton, soybeans and corn reduced soil erosion by 50, 65 and 84% respectively, compared to conventional tillage practices.

Surface runoff can be reduced by up to 99% through conservation tillage practices (Edwards et al. 1988, Fawcett, 1994). Cracks, roots, channels, and wormholes create macropores that increase the infiltration capacity of the soil in the absence of tillage. During storm events, 1 to 10% of rainfall flows away from the surface through these holes. Burrows and large soil pores are destroyed in tilled and altered soils (Edwards et al. 1989). A study conducted in the silt loam region of the Dawson Creek, British Columbia found infiltration was 60% greater with no-till than conventional tillage (Arshad et al. 1999; Meyers et al. 2003).

Like conservation tillage, permanent cover plays a major role in reducing soil erosion. The CRP has reduced erosion by more than 22% in the United States even though less than 10% of cropland is enrolled (Ribaudo et al. 1990). Davie and Lant (1994) studied the CRP in two watersheds in Illinois and found results similar to the national average with a 24% and 37% erosion reduction in each watershed. The CRP has also created 13,600 km of buffer strips (i.e., vegetated areas around wetlands and along watercourses) and 700,000 ha of grassland habitat.

Blackburn et al. (1991) determined annual runoff decreased on almost all CRP sites studied and attribute this to increased evapotranspiration, greater infiltration and improved site stability from perennial grass. Litter cover and root biomass, provided by perennial grass, protects soil throughout the year from the erosive forces of rain, runoff and wind. In Mississippi, Gilley and Doran (1997) found that when simulated rainfall was applied to nearly saturate soils, there was minimal runoff from CRP sites.

Tillage often destroys soil aggregation and structure resulting in increased potential for wind and water erosion (Karlen et al. 1998). Studies indicate the CRP and no-till have effectively reduced runoff because the soil contained a greater proportion of stable aggregates, which improved soil structure (Karlen et al. 1999; Weinhold and Tanaka 2000; Elliott et al. 2001; Huang et al. 2002).

The projected reductions in erosion and the US dollar savings resulting from water quality improvement from land enrolled in CRP are shown in Table 7 (Ribaudo 1989; Ribaudo et al. 1990).

**TABLE 7 – Projected reduction in erosion, nutrient discharge and the US dollars savings resulting from water quality improvement from land enrolled in CRP.**

| Erosion Reduction | | | Total Suspended Solids | | Total Kjeldahl Nitrogen | | Total Phosphorus | | U.S. Dollars |
|---|---|---|---|---|---|---|---|---|---|
| Regions by States | Area: 1000 ha | Soil Saved million kg | million kg | % reduced | 10,000 kg | % reduced | 10,000 kg | % reduced | Saved (millions) |
| IA; MO; IL; IN; OH | 3,095 | 123,551 | 62,668 | 12.0 | 25,309 | 12.1 | 4,816 | 11.9 | 746 |
| MN; WI; MI | 1,533 | 29,872 | 10,490 | 14.2 | 6,066 | 13.1 | 560 | 11.4 | 415 |
| PA; NY; MD; NJ; CT; MA; NH; ME | 295 | 8,230 | 3,729 | 3.7 | 1,666 | 2.8 | 406 | 3.0 | 191 |
| ND; SD; NE; KS | 3,897 | 71,530 | 34,525 | 11.0 | 12,508 | 14.4 | 2,560 | 13.3 | 267 |

### c) Nutrient Assimilation

#### *1. Nitrogen*

Nitrogen is the fertilizer nutrient applied in the greatest quantity and is most limiting with respect to crop production (Weinhold and Tanaka 2000; Malhi et al 2001). Nitrogen is lost mainly through erosion, leaching, denitrification, and volatilization with erosion and leaching being the two that can degrade water quality.

Studies have shown that conservation tillage can reduce nitrogen loss through runoff. A 97% reduction in sediment loss from no-till resulted in a 75 to 90% reduction in total nitrogen loss for soybeans planted following a corn crop (Baker and Laflen 1983). Regardless of the tillage system, Seta et al (1993), found nitrate concentrations in runoff water were associated with rainfall events. They found nitrate concentrations from no-till were 157 and 139% higher than chisel-plow and conventional tillage, respectively, but total nitrate loss was 71 and 86% less for no-till because of reduced runoff volume from surface residue.

Even though no-till can effectively reduce surface losses of nitrogen, the amount lost through leaching is less clear. It is already known that macropores can increase water infiltration capacity of the soil in the absence of tillage (Edwards et al. 1988; Elliott and Efetha 1999) so some fear no-till might also increase the leaching of chemicals through the soil. Malhi et al. (2001) reviewed nitrogen fertilization for no-till cereal production in the Canadian Great Plains and reported nitrate is subject to loss from leaching because it is more mobile in solution.

A number of studies indicate no-till had little impact on nitrate leaching (Kanwar et al. 1988; Kanwar and Baker 1993; Randal and Iragavarapu 1995; Elliott et al. 2001) while other research found an increase in nitrogen loading when no-till was used (Sharpley and Smith 1994; Izaurralde et al. 1995). Randal and Iragavarapu (1995) and Rasse and Smucker (1999) found more water was lost through leaching in no-till compared to conventional tillage but the nitrate concentrations were greater with conventional tillage.

With respect to permanent cover, a study conducted by Randall et al. (1997) determined that substantially lower nitrate concentrations were found in drainage from alfalfa and CRP (planted with a mixture of alfalfa, smooth bromegrass (*Bromus inermis L.*), orchardgrass (*Dactylis glomerata*) and timothy (*Phleum pratense*)) compared with continuous corn cropping and a corn-soybean rotation. Nitrate lost during the 4-year study was 218 kg/ha for continuous corn cropping, 203 kg/ha for corn/soybean rotations, 7.2 kg/ha for alfalfa, and 4.5 kg/ha for CRP. High nitrate concentrations, in addition to high drainage volumes, resulted in nitrate losses being about 45 times higher in row crops compared to perennial crops.

In Idaho, Robbins and Carter (1980) found lower nitrate concentrations in drainage water with growing alfalfa compared to dry bean and corn. Since alfalfa effectively reduces nitrate concentrations in soil, it could be used as a management tool for removing nitrate from the soil below the rooting depth of most crops (Russelle and Hargrove 1989).

Karlen et al. (1996) researched soil quality of CRP sites planted to fescue (*Festuca* sp.) and smooth bromegrass. They determined nitrate concentrations of CRP soils were significantly lower than those found at cultivated sites. They presume this is the result of longer growing seasons for grass and its ability to assimilate any available nitrate. Another study conducted by Karlen et al. (1999) found that nitrate levels were significantly (18-74%) higher in cropland sites than CRP in Iowa, Minnesota, North Dakota and Washington. It was concluded that input of fertilizer nitrogen was the cause, although tillage can increase soil nitrate concentrations compared to grasslands.

Table 7 illustrates the estimated reductions in total nitrogen and savings resulting from water quality improvement from land enrolled in CRP (Ribaudo 1989; Ribaudo et al. 1990).

*2. Phosphorus*

Phosphorus is the fertilizer nutrient that aids in accelerated eutrophication of receiving waters (Sharpley et al. 2000); consequently, a reduction in phosphorus loading by using upland conservation practices should result in improved water quality. Phosphorus loss in surface runoff can be attached to sediment or in a dissolved form. Surface runoff is often the leading pathway of phosphorus loss from agriculture although loss through soil percolation can occur to a lesser extent (Heathwaite et al. 2000). The concentration of phosphorus lost through percolation is usually small because phosphorus is absorbed by phosphorus-deficient subsoils (Heathwaite et al 2000; Sharpley et al. 2000).

Even though phosphate concentrations in runoff from no-till were significantly higher (222%) than conventional tillage, total phosphate loss from conventional tillage was extremely high (248%) compared to no-till (Seta et al. 1993). They attribute higher concentrations of phosphate to lower runoff volume and to crop stubble intercepting fertilizers. Andraski et al. (1985) determined no-till reduced total phosphate loss through runoff by 81% compared to moldboard plow. Baker and Laflen (1983) found a similar result with an 80 to 91% reduction in phosphorus loss when soil erosion was reduced by 97% through the no-till conservation practice.

Other research has shown lower reductions in phosphorus loading by using no-till. In the effort to reduce eutrophication in Lake Erie, conservation tillage practices were implemented. No-till, in association with reduced fertilizer applications, led to a reduction in total phosphorus loading of 24% (Richards and Baker, 1993). In the Mississippi Delta region of the United States, Intarapapong et al (2002) discovered no-till reduced phosphorus loss in sediment by 26.6% for soybeans and 46.7% for corn as compared to conventional tillage practices. A study conducted in Oklahoma by Sharpley and Smith (1994), determined total phosphorus loss from surface runoff was reduced by 70% as the result of no-till.

With respect to permanent vegetative cover, Huang et al. (2001) studied soil properties along a transect that was partially CRP land and partially continuously cropped land in Central Kansas. They found concentrations of available phosphorus significantly lower in CRP compared to the continuously cropped land. The study also reports that the accumulated phosphorus in the root zone of continuously cropped land is likely the product of intensive fertilizer applications. As a result of the CRP, Weitman (1994) determined phosphorus loadings were reduced by approximately 30% in some regions of the United States.

Table 7 shows the expected reductions in total phosphorus and the US dollar savings resulting from water quality improvement from land enrolled in CRP (Ribaudo 1989; Ribaudo et al. 1990).

### *d) Pathogens*

Research regarding pathogen movement and upland conservation practices is limited. One aspect of a new study, conducted by Ohio State University and North Carolina State University, is comparing the movement of pathogens through soil macropores for no-till and conventional tillage practices (Ohio State University 2003). Information is expected at the end of this four-year study in 2006.

Some information is currently available concerning macropores and microbial movement through soil. Smith et al. (1985) compared intact soil to disturbed soil and found that 93% of the inoculated *E. coli* was retained in disturbed soil; whereas intact soil retained 21-73% of inoculated cells. Saini et al. (2003) studied leaching potential of fecal pathogens from swine manure application. They tested three manure application methods (no-till broadcast, broadcast on tilled soil and incorporation following broadcast) and discovered there was no difference with regard to leaching, although there was greater pathogen survival when the manure had been incorporated into the soil.

One major factor influencing microbial transport in soil is soil type. In a study conducted by Patni et al. (1984), it was determined that pathogen dissemination has greater potential in coarsely textured soils with large pore spaces than in finer textured, less porous soil. Mawdsley et al. (1995) concluded that agricultural practices influence pathogen dissemination depending on the type of waste distributed, the degree of soil disturbance and the presence of root crops.

Inadequate information is available concerning the effects of upland cover and overland pathogen movement. It is reasonable to assume that pathogen movement through upland cover would be similar to that of other chemicals and nutrients, and that corresponding management techniques would be beneficial, but without sufficient data, this cannot be concluded. Until further research is completed, pathogen contamination of groundwater and surface water can be regulated through proper manure storage, controlled livestock watering, manure application methods, timing and restricted application to soils that are prone to runoff (Hilliard et al. 2002; Agriculture and Agri-foods Canada 2003).

### *e) Pesticides*

In Canada, 81% of pesticides applied to agricultural crops were herbicides (Paoletti 1997). Pesticides may be transported to the environment or atmosphere by volatilization, sprayer drift, wind erosion, water runoff and erosion and leaching (Hall et al. 1989).

Like nitrogen and phosphorus, pesticide runoff is usually reduced with conservation tillage practices. Baker et al. (1995) reviewed published studies from 1990 to 1995 that investigated herbicide runoff from no-till and found a 70% reduction in herbicides. Buffer strips added to minimum till practices further reduced herbicides in runoff by an average of 46%. They stated that the effectiveness of surface crop residue in reducing herbicide runoff depends on the site and weather conditions. Other studies found no-till completely eliminated pesticide runoff due to complete water infiltration (Foy and Hiranpradit 1989; Glenn and Angle 1987; Hall et al. 1991).

With regard to pesticide leaching in no-till fields, data are conflicting. Some research has reported increased leaching of pesticides (Hall et al 1989; Isensee et al. 1990) while others found the reverse is true (Fermanich and Daniel 1991; Levanon et al. 1993; Gish et al. 1995; Isensee and Sadeghi 1996; Stoltenberg et al. 1997; Novak 1997). Herbicide leaching appears to be less of a problem in semi-arid Canadian prairies than in mid-western United States because cropping systems differ. On the Canadian Prairies, the majority of herbicides (bromoxynil, 2,4-D, and dicambia) are applied post emergence at low use rates, and they have a shorter persistence in the environment (Derksen et al. 1996).

Despite the fact that pesticide concentrations in runoff may be high in certain fields, reductions in total mass of pesticide runoff by conservation tillage practices should reduce concentrations of pesticides in many wetlands, rivers, and lakes (Fawcett et al. 1994). Even though there is an increased potential for pesticide leaching with no-till, the relative concentrations found in surface water are typically greater than those found in groundwater (Uri 2000).

Ribaudo et al. (1990) felt the CRP would probably not significantly affect national groundwater quality although it could improve groundwater quality in several regions of the United States. Approximately 65% of the cropland vulnerable to groundwater pollution in the Southern Plains and 31% in the Mountain region are eligible for CRP enrollment. If these areas are removed from agricultural production through the CRP, this land will have lower rates of pesticides and fertilizer nutrients applied and subsequently released into the groundwater (Ribaudo et al. 1990). Huang et al (1990) came to a similar conclusion. They suggested cropland overlying groundwater that is susceptible to pesticide contamination should be placed into the CRP. Weitman (1994) also suggested herbicide loadings in the United States were reduced by 50% because of the CRP.

### *f) Summary*

Upland conservation programs, such as no-till and permanent perennial cover, slow surface runoff, trap sediments and promote infiltration, consequently reducing the amount of sediments, nutrients and pesticides that enter the watershed.

The most beneficial outcome from implementing practices such as conservation tillage and permanent perennial cover is erosion reduction (Table 8). Erosion from wind, rain and runoff is reduced significantly because of greater site stability, infiltration and protection as a result of surface crop residue and perennial vegetation.

**TABLE 8 – Range of percent retention for nitrogen, phosphorus, sediment, coliforms and pesticides in natural wetlands.**

| | Reduction (%) | |
|---|---|---|
| | *No-till* | *CRP* |
| Sediment | 50-99 | 22-37 |
| Nitrogen | 71-90 | 2.8-14.4 |
| Phosphorus | 24-91 | up to 30 |
| Pesticides | 70-100 | up to 50 |

Upland cover has shown to be effective in reducing the amounts of nitrogen, phosphorus and pesticides in runoff (Table 8) but there is potential for increased leaching through the soil profile to groundwater. Although conservation tillage has not always reduced nutrient and pesticide leaching, this practice is recommended because the benefits outweigh the potential drawbacks. Research indicates conservation tillage is effective but it should be established on an individual farm basis. With respect to perennial upland cover, this land has been removed from production, resulting in fewer pesticides and fertilizer nutrients applied and subsequently released (Table 8). Currently there is insufficient information regarding upland conservation practices and pathogen movement.

A combination of upland practices in conjunction with fertilizer and pesticide management should help reduce the amounts of nitrogen, phosphorus, pathogens and pesticides in runoff and leachate. The implementation of BMPs has been accepted as the most feasible solution for reducing non-point source pollution from agriculture, which in turn will help improve water quality.

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