FINAL REPORT

NATURAL ATTENUATION OF NITROGEN IN WETLANDS AND WATERBODIES

To:
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April 2007

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COMMONWEALTH OF MASSACHUSETTS
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MassDEP SYNOPSIS

The Massachusetts Department of Environmental Protection (MassDEP) is pleased to provide you, the readers of this report, with the findings of this MassDEP/USEPA funded study. MassDEP commissioned this search of the literature by the contractor, the Woods Hole Group and its subcontractor, Teal Partners, to provide MassDEP policy and regulatory staff and the Massachusetts Estuaries Project (MEP):

- The current state of our knowledge concerning the attenuation/cleanup of nitrogen contaminated ground and storm water by both natural and constructed wetlands
- The effectiveness of natural wetland system processes for removing nitrogen contaminated groundwater by wetland ecosystems
- Optimal designs and site modifications of wetlands to enhance nitrogen removal by natural attenuation
- What the literature has reported concerning the benefits and detriments of nitrogen attenuation on wetland ecosystems
- MassDEP data needs for the review of natural attenuation project proposals

The contractor reviewed over 200 articles and reports about natural attenuation of nitrogen in different types of wetlands (bogs, fens, emergent, shrub-scrub, wet meadows, cranberry bogs, forested & open wetlands, salt ponds, marshes and mudflats) and waterbodies (streams, rivers, lakes and ponds). Information was also sought from the researchers who have authored previous studies for any unpublished/in press studies. Publications were also sought on the design for constructed wetlands and the site modifications to enhance natural attenuation rates. Finally, the literature review also examined data obtained from model, laboratory, and field projects.

This review of the literature identifies denitrification in wetlands as the most effective nitrogen removal mechanism from surface and ground water, followed in effectiveness by small ponds, large ponds and streams. Vegetative uptake played only a minor role in nitrogen removal. The role of pH, oxygen content, muck content as a carbon source, stream and/or groundwater flow, and temperature are fully described, each with optimal environmental conditions for promoting nitrate attenuation.

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Following the completion of this literature review, the contractor, as a contract deliverable, presented its findings at two public forums on April 24, 2007 at the Buttonwood Park Zoo in New Bedford and on April 25, 2007 at the Harwich Community Center. These meetings were well attended and strategically important to the Department and the MEP in providing the public's point of view on the use of natural and enhanced nitrogen attenuation processes.

This research represents a first step in the policy development process for external and internal discussion concerning the effectiveness, limitations in use, and applicability under existing state statutes and regulations of nitrogen attenuation. The findings of this review of the literature will allow the MassDEP to consider the effectiveness of nitrogen attenuation as a treatment option to reduce impacts from nitrogen-contaminated groundwater that would otherwise contribute to estuarine eutrophication.

Glenn Haas
Acting Assistant Commissioner for Resource Protection

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MassDEP ADDENDUM

The Massachusetts Estuaries Project (MEP) has investigated natural attenuation percentages in several embayments including those in Chatham (Howes, *et al.,* 2003), Mashpee (Howes, *et al.,* 2004 and Howes, *et al.,* 2005a), Falmouth (Howes, *et al.,* 2005b), Barnstable (Howes, *et al.,* 2006a and Howes, *et al.,* 2006b), Falmouth (Howes, *et al.,* 2006c and Howes, *et al.,* 2006e) and Pleasant Bay (Howes, *et al.,* 2006d). Nitrogen removals were calculated for pond, stream, pond/stream and salt marsh systems.

MEP findings for freshwater systems indicated an approximate range of 35%-95% with lower removals found in stream systems and greater attenuation in ponds. Attenuation percentages in the freshwater systems were determined in one of two ways. Both methods compared predicted input nitrogen loads, as calculated from a detailed land-use-loading model, to nitrogen output. One method determined nitrogen output based on nitrogen concentrations in ponds collected during the summer and then calculating nitrogen mass output based on discharge volume and turnover time. Using this pond survey method, Howes *et al.* report attenuation percentages between 39%-95% in several ponds in Chatham, Massachusetts (Howes *et al.*, 2003) and attenuation percentages of 84%-96% in ponds in the Centerville River/East Bay (Massachusetts) watershed (Howes *et al.*, 2006b). However, one must be cautious in use of these attenuation factors because the in-pond sampling data is limited to the summer season and may not be representative of the annual load.

The second method directly calculated nitrogen output by measuring streamflow and nitrogen concentrations in streams located at the outlet end of a pond system. Here, attenuation percentages were integrated as nitrogen passed through ponds and streams and demonstrated attenuation percentages between as low as 22% with upper values approaching 70% (Howes *et al.,* 2006a). Lower attenuation percentages in these pond/stream systems are probably accounted for by the hydraulic behavior of the systems as shallow flow through systems with limited detention time. In systems that were predominantly riverine in nature, attenuation percentages of 30% to 40% were observed (Howes, *et al.,* 2005a).

MEP reports also evaluated attenuation in salt marshes (Howes, *et al.*, 2003, Howes, *et al.*, 2006e and Howes, *et al.*, 2007). Attenuation percentages reported based on previous work range between 40% and 50%.

The MEP uses general attenuation factors of 50% for ponds, 30% for streams and 40% for salt marshes for systems where site-specific information is not available. These factors fall within observed ranges and provide an overall degree of conservatism. While these rates are calculated using both models and *in situ* measurements, it is beyond the scope of the MEP to evaluate attenuation mechanisms or processes.

References Cited

Howes, B., S. Kelley, J. Ramsey, R. Samimy, D. Schlezinger, and E. Eichner. (2003). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Stage Harbor, Sulphur Springs, Taylors Pond, Bassing Harbor, and Muddy Creek, Chatham, Massachusetts. Commonwealth of Massachusetts, Department of Environmental Protection, Massachusetts Estuary Project, 272 pp. Boston, MA.

Howes, B., S. Kelley, R. Samimy, E. Eichner, D. Schlezinger, and J. Wood. (2004). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Popponesset Bay, Mashpee and Barnstable, Massachusetts. Commonwealth of Massachusetts, Department of Environmental Protection, Massachusetts Estuary Project, 138 pp. Boston, MA.

Howes B., S.W. Kelley, J.S. Ramsey, R. Samimy, D. Schlezinger, T. Ruthven, and E. Eichner (2005a). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Quashnet River, Hamblin Pond, and Jehu Pond, in the Waquoit Bay System in the Towns of Mashpee and Falmouth, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes, B., J., S. Ramsey, S.W. Kelley, R. Samimy, D. Schlezinger, and E. Eichner (2005b). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Great/Perch Pond, Green Pond, and Bournes Pond, Falmouth, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA., 205 pp.

Howes B., H.E. Ruthven, J. S. Ramsey, R. Samimy, D. Schlezinger, J. Wood, and E. Eichner (2006a). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Centerville River System, Barnstable, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B., S. W. Kelley, J. S. Ramsey, R. Samimy, D. Schlezinger, and E. Eichner (2006b). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Three Bays, Barnstable, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes, B. L., J. S. Ramsey, E. M. Eichner, R. I. Saminy, S. W. Kelley, and D. R. Schlezinger, (2006c). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Little Pond System, Falmouth, Massachusetts. SMAST/DEP Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes, B., S. W. Kelley, J. S. Ramsey, R. Saminy, D. Schlezinger, and E. Eichner (2006d). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Pleasant Bay, Chatham, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes, B., S. W. Kelley, J. S. Ramsey, R. Saminy, D. Schlezinger, and E. Eichner (2006e). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for West Falmouth Harbor, Falmouth, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B.L., E.M. Eichner, S. W. Kelley, J. S. Ramsey, and R.I. Samimy (2007). Water-Use Update to the Linked Watershed-Embayment Model to Evaluate Critical Nitrogen Loading Thresholds for Stage Harbor/Oyster Pond, Sulphur Springs/Bucks Creek, Taylors Pond/Mill Creek, Chatham, Massachusetts. Massachusetts Estuaries Project Technical Team Update Report.

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ABSTRACT

We reviewed nearly 200 articles and reports related to natural attenuation of nitrogen in different types of wetlands (bogs, fens, emergent, shrub-scrub, wet meadows, cranberry bogs, forested & open wetlands, salt ponds, marshes and mudflats) and waterbodies (streams, rivers, lakes and ponds). We also reviewed the literature on design for constructed wetlands and reviewed articles that described site modifications that enhance natural attenuation rates. The literature review examined data obtained from model, laboratory, and field projects. The literature indicated that the most effective nitrogen removal from surface and ground water is via denitrification in wetlands, small ponds, large ponds and streams. Vegetative uptake played only a minor role in nitrogen removal. The most important physical characteristics of the wetland or water body that enhanced nitrogen removal are nitrate loading, detention time, anoxic zones, organic carbon, temperature and pH. Specifically, conditions that maximize nitrogen removal include a nitrate loading rate of ~ 2 to 3 mg/l, detention time of about one day in anoxic zones with labile organic carbon, near neutral pH, and temperatures ~ 10° C. We also described the role of climate (wind, rain, season, air and water temperature). Finally, we described wetland modifications that may enhance nitrogen removal from ground and surface waters in Massachusetts.

EXECUTIVE SUMMARY

Massachusetts has a solid history of wetland protection that recognizes the broad range of functions these systems provide. These wetland systems are often balanced precariously between the uplands and a water body such as river, lake, estuary, or an ocean. While these wetlands are often viewed as fragile, in reality they are quite durable provided they are not physically altered and can sustain their existence because they have adapted to the transitional environment. These systems provide protection for many species of flora and fauna; they provide breeding, nesting, and nursery habitat for many species. They also provide important non-quantifiable functions such as recreational benefits, areas for research and educational programs. Some of the wetlands are valued for their ecological functions, others for their societal functions, and some for both. One of the functions not specifically listed in Massachusetts Wetland Protection Act is the ability of wetlands, ponds, streams and lakes to attenuate nitrogen. As inland and coastal waters have become more polluted with nitrogen we have learned there are negative effects on both the ecosystem and society.

To counter these negative impacts, both federal and state governments have initiated the Total Maximum Daily Loading (TMDL) program. The Clean Water Act, Section 303, establishes the water quality standards and TMDL programs. A TMDL load is a calculation of the maximum amount of a pollutant that a waterbody can receive and still meet water quality standards, and an allocation of that amount to the pollutant's sources.

States, Territories, and Tribes set water quality standards. They identify the uses for each waterbody, for example, drinking water supply, primary contact recreation (swimming), secondary contact recreation (boating), and aquatic life support (fishing), and the scientific criteria to support that use. A TMDL is the sum of the allowable loads of a single pollutant from all contributing point and nonpoint sources. The calculation must include a margin of safety to ensure that the waterbody can be used for the purposes the State has designated. The calculation must also account for seasonal variation in water quality. Nitrogen is considered a primary pollutant under the TMDL program.

The Massachusetts landscape is a mosaic of wetland types that use and transform nitrogen. This report summarizes nitrogen processing functions for each wetland type and considers whether nitrogen attenuation functions (particularly denitrification) could be enhanced, without damage to a wetland or waterbody, to meet society's goal of reducing nitrogen loading to the coastal ocean.

We conclude that natural nitrogen attenuation projects can be designed and implemented such that the high level of nitrogen (nitrate) carried by stream, rivers, and estuaries can be artificially introduced into some wetlands and waterbodies such that the excess nitrate will be denitrified efficiently with low amounts taken up by plants, stored in sediment or lost to outflow. This discussion does not deal with ammonia since, while nitrate is so soluble that it moves with water flows as if it were water, ammonia attaches to sediments and moves very little until it is oxidized to nitrate.

1.0 INTRODUCTION

The purpose of this review for the Massachusetts Department of Environmental Protection and the Massachusetts Estuaries Project (MEP) is to document the effectiveness of natural attenuation of nitrogen in different types of wetlands and waterbodies, describe designs and site modifications to enhance existing natural attenuation rates, and list data needs for review of natural attenuation project proposals. Enhanced natural attenuation of nitrogen, in combination with wastewater and stormwater management, may reduce N loading from a watershed to the coastal ocean.

We reviewed 183 published scientific papers and gray literature that assess nitrogen retention or attenuation in freshwater and saltwater wetlands (bogs, fens, emergent, shrub-scrub, wet meadows, cranberry bogs, forested & open wetlands, salt ponds, marshes, mudflats and constructed wetlands) and waterbodies (streams, rivers, lakes and ponds). We focused on geographic areas with characteristics (hydrology, geology, climate, growing seasons, etc.) similar to coastal Massachusetts. From the papers we reviewed, we tabulated wetland/waterbody information relevant to nitrogen retention/attenuation in each article. This information is found in **Appendix A** (provided in electronic format).

The tabulation includes the following:

- Physical characteristics: size, water depth and volume, sediment volume, depth, organic content and grain size, stream sinuosity
- Chemical characteristics: redox potential of sediments; air and water temperature; salinity, water quality (DO, BOD, nutrients, pathogens, presence of other contaminants)
- Biological characteristics: vegetation and wildlife types, abundance, and densities; potential for algal blooms and eutrophication; seasonality of vegetation
- Processes and process-related variables: wind levels; sunlight, groundwater and surface water flows; tidal hydrodynamics; flushing rates; residence time.

The tabulation shows that none of the articles had complete information about the physical, chemical, biological and environmental features of the wetlands or waterbodies described. It does provide other researchers with guidance as to which articles may be useful for their specific data needs. This preliminary review allowed us to select a subset of articles for detailed review. **Appendix B** contains bibliographic information for each article, including the published abstract (if available). For the articles most relevant to this discussion, an annotation or summary is provided. Appendix B is organized by wetland or waterbody type so that one may easily see the number of articles, type of information, and range of information used to generate the text below. Rather than referring to all annotated articles in the text describing the various wetland types, we refer readers to the appropriate section of **Appendix B**. There are specific instances in the text where we provide data; those are in standard scientific notation and included here as Cited References.

Required Reading. There are 10 articles that should be read by all. Seitzinger (1988) provides a comprehensive literature review of denitrification in freshwater and coastal marine ecosystems. The other nine papers were published in Ambio (the Journal of the Royal Swedish Academy of Sciences) in September 1994. Jansson et al. (1994a) described how adverse effects of increased nutrient loading began to be noticed in the 1960s, and that there were severe impacts by the 1980s with "algal blooms, expansion of reduced bottom areas with sulfide production and partial extinction of fish and crustacean populations" in the Baltic Sea. The response of Sweden and neighboring countries was to support programs to reduce nitrogen discharge by half by 1996. [It should be noted that this goal was not met.] The measures they proposed were to improve sewage treatment for municipal and industrial discharges, reduce nitrogen inputs from agriculture and forestry, and reduce atmospheric deposition. This led to the studies carried out under a research program: Wetlands and Lakes as Nitrogen Traps reported in the special issue of Ambio.

While these articles cover only fresh water systems, everyone involved in technical review of natural attenuation proposals should read them. One of their conclusions important to Massachusetts nitrogen management policy, is:

If wetlands and waterbodies covered ~1% of the surface area of a southern Sweden subwatershed, one could expect < 15% of the nitrogen would be attenuated or retained (Jansson et al. 1994a). Using models specific to their watershed, Arheimer and Wittgren (1994) estimated that with ~5% of the watershed as wetlands or waterbodies, one could expect ~50% reduction of the nitrogen reaching the coastal ocean in southern Sweden.

The sources of nitrogen in southern Sweden were estimated to be 50% from agriculture and 30% from the atmosphere. While coastal Massachusetts communities have some agriculture, the major nitrogen sources in this area include those that result from residential development: on site sewage systems and lawn fertilizer. Nevertheless, the nitrogen retention processes are the same, and these important papers centered on southern Sweden wetlands are applicable to Massachusetts wetlands.

2.0 NITROGEN ATTENUATION (RETENTION) AND NITROGEN CYCLING IN WETLANDS

2.1. THE NITROGEN CYCLE

Key processes related to nitrogen retention or attenuation in wetlands include nitrification, denitrification, and plant uptake. Denitrification is the most important process because it involves a loss of nitrogen from the wetland back into the atmosphere as nitrogen gas, rather than temporary storage in living or dead plant tissue (Weisner et al. 1994, Fleischer et al. 1994, Jansson et al. 1994a, Jansson et al. 1994b). These processes take place on different temporal and spatial scales, and wetland retention rates are limited by temperature, pH, redox, available carbon, DO, and, of course, nitrogen. As mentioned in the Executive Summary, this discussion deals with nitrate rather than ammonia because ammonia attaches to sediments and moves very little until it is oxidized to nitrate.

Figure 1 shows the nitrogen cycle but does not show the complex biogeochemical functions that transform organic nitrogen to ammonium (NH_4^+), to nitrite (NO_2^-), to nitrate (NO_3^-), to atmospheric nitrogen (N_2) and back around the circle again.

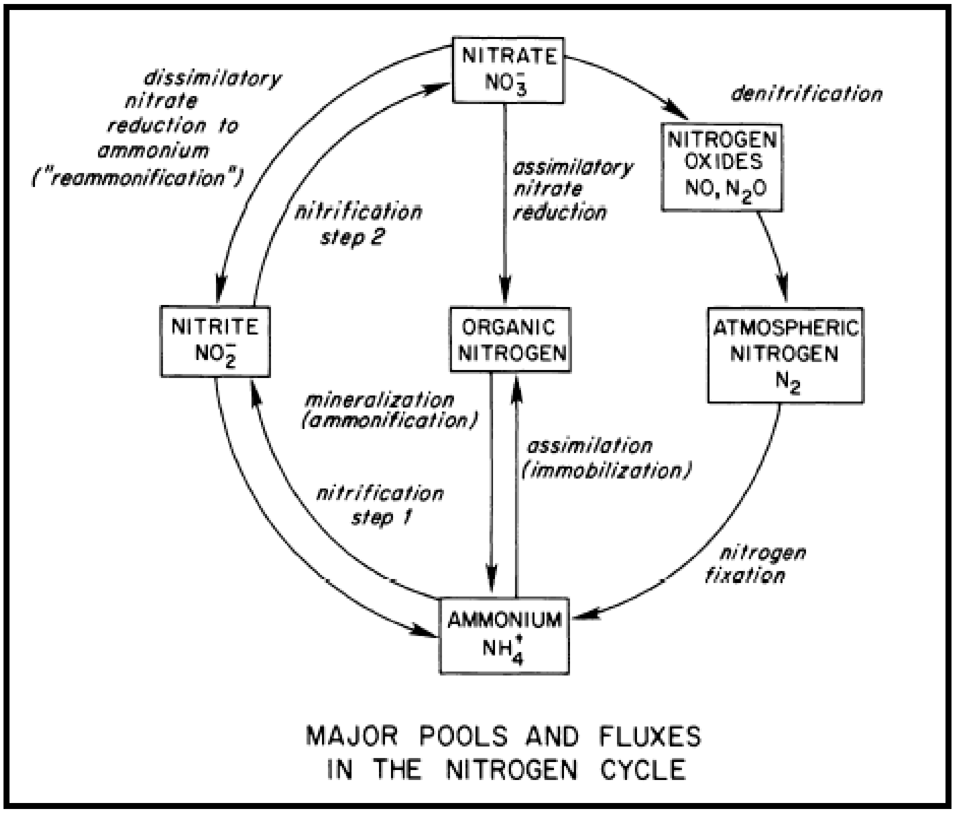

Figure 1. Schematic of the nitrogen cycle, after Bowden 1987

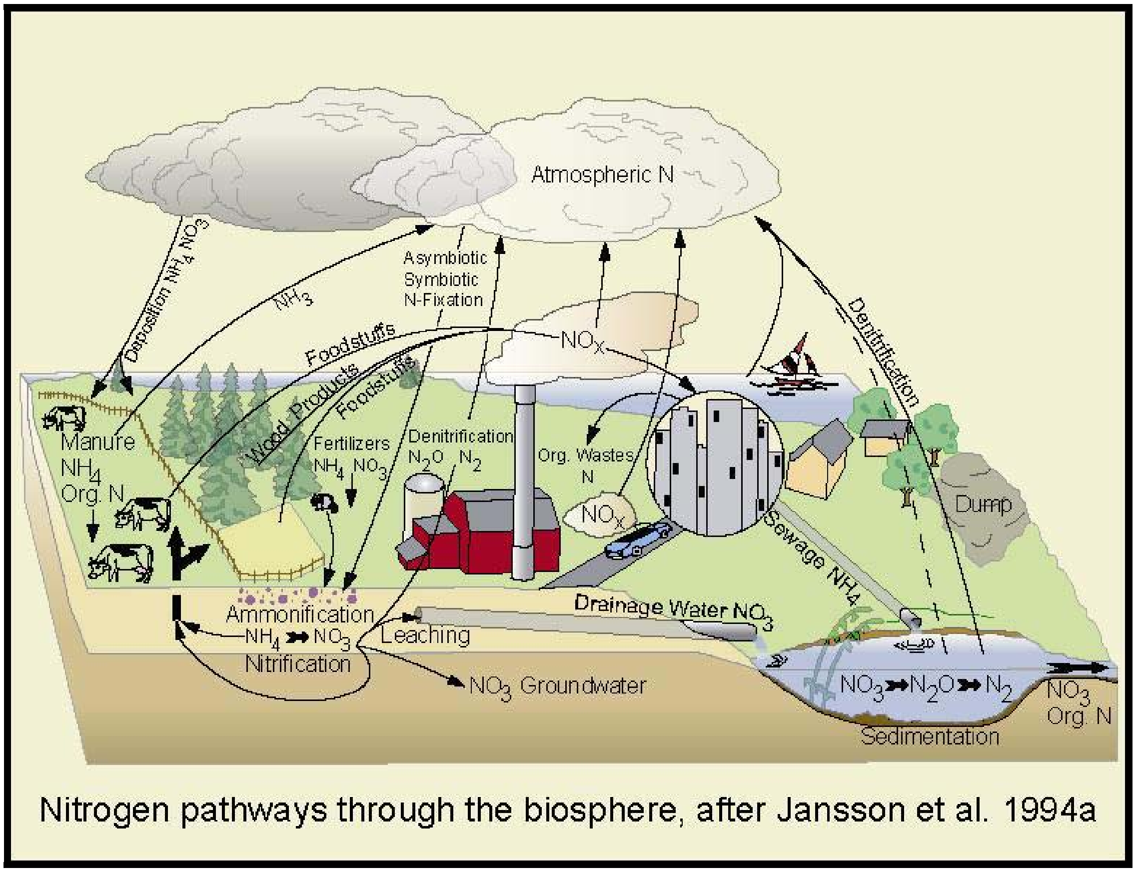

Figure 2. Nitrogen pathways through the biosphere, after Jansson et al. 1994a

Figure 2 helps to explain what happens to nitrogen in earth's environment. N_2 makes up 80% of earth's atmosphere. Some plants "fix" nitrogen, taking it from the air and using it for tissue generation. There is another form of nitrogen in the air, NO_X or nitrous oxides, which are combustion products. NO_X are deposited on the earth's surface with rain. Other forms of nitrogen – organic nitrogen -- enter ground and surface waters as decay products or waste streams. Some organic nitrogen is converted to ammonium by bacteria in soils and water. Not all organic N is readily converted to ammonium; we generally call this recalcitrant organic N because it takes so long for the N to be released. Thus it does not contribute significantly to eutrophication in coastal waters. Ammonium is taken up by plants and stored in roots, rhizomes, nodules, woody parts and leaves. Some is released as leaf fall; the rest is released when plants die. Nitrogen in herbaceous plants and leaves is recycled annually in Massachusetts. Nitrogen in wood takes longer to be recycled – perhaps 10 to > 100 years. Ammonium, when released into the environment by decay, is used as an energy source by bacteria in environments with low organic matter and temperatures above about 5^oC and is converted to nitrate, a process called nitrification.

Nitrification generates NO_2^- (nitrite) that is further oxidized to NO_3^- (nitrate). Plants can also take up nitrate. The more important process for nitrogen attenuation is denitrification. This occurs in anoxic environments when bacteria use the oxygen in nitrate for respiration and release N gas back to the atmosphere. This is less temperature sensitive than nitrification, but increases about 2 to 3-fold with every 10°C rise in temperature (between < 4° and 40° C) (Jansson et al. 1994a). This process requires labile carbon (dissolved organic matter). In sewage treatment plants, methanol is a source of carbon, as is sugar, sodium acetate, or Micro-C®. In natural systems, the labile carbon is most often supplied by decaying plants and other organic matter.

In some environments, such as wetlands, shallow ponds, stream bottoms, and on the surfaces of submerged plants, nitrification and denitrification co-occur. This happens because there are oxic and anoxic microenvironments in these habitats.

2.2. NITROGEN REMOVAL/RETENTION/ATTENUATION

This following section focuses on nitrate, the dominant form of nitrogen entering wetlands and waterbodies, and on microbial denitrification as the most important process for reduction of N loading to the environment[1]. Weiskel and Howes (1992) provide an excellent discussion of natural attenuation of nutrients as they flow with the discharge from septic systems through soil on the way to groundwater. They report that, with adequate infiltration distance through sandy soils where oxidizing conditions occur or there is free molecular oxygen, the chemical changes in the groundwater flowing through the soil were ~70% nitrification of ammonium and ~60% retention of phosphate by the soil. The same processes occur for storm water and rainwater as for septic system effluent.

Seitzinger (1988) provides a good characterization of the process and controls on denitrification, and the ranges of denitrification rates in various environments. In general, the Seitzinger review discusses three types of wetland systems that pertain to Massachusetts: 1) rivers/streams, 2) lakes, and 3) estuaries (coastal marine). These systems commonly experience denitrification rates ranging from 0-345 µmol N m^{-2} h^{-1} in rivers/streams, 2-171 µmol N m^{-2} h^{-1} in lakes, and 50-250 µmol N m^{-2} h^{-1} in coastal marine ecosystems. A denitrification rate range and annual average for Narragansett Bay was 39-109 [59] µmol N m^{-2} h^{-1}.

The nine articles published in *Ambio* in September 1994 (fresh water systems) were in response to Swedish public policy calling for 50% reduction of N loading to the coastal environment. They analyzed the effectiveness of wetlands, ponds, lakes, forested wetlands, riparian systems, streams, created ponds, and artificially flooded meadows to determine if any or all retained N. The most effective systems were shallow ponds with down gradient wetlands. Annual net retention in these systems varied between 73 and 7000 kg N ha^{-1} (Fleischer et al. 1994). One of their conclusions is that if wetlands and waterbodies covered ~1% of the surface area of a subwatershed, one could expect < 15% of the nitrogen would be attenuated or retained by these systems (Jansson et al. 1994a). Using models specific to their watershed, they estimated that with ~5% of the watershed

[1] Some of the literature we reviewed described field studies of wetlands and waterbodies where nitrification did not occur – the nitrogen cycle didn't get started; that literature was not useful for our evaluation of attenuation.

as wetlands or waterbodies, one could expect ~50% reduction of the nitrogen reaching the coastal ocean (Arheimer and Wittgren 1994).

Note that there are several ways to express nitrogen concentration. In water quality testing, we often see the data reported in mgl^{-1} (weight per unit liquid); this is useful when dealing with a known volume of water. But for natural systems such as wetlands, ponds and the coastal ocean, this notation is not as useful. Another, and more useful expression for our purposes, is molar notation. A mole is the quantity of a substance whose mass in grams is the same as its atomic weight. This allows apples to apples comparisons among the various N forms. The term "µmol N m^{-2} h^{-1} "should be read as "micromoles of nitrogen per square meter per hour".

Note that several terms are used interchangeably for nitrogen removal from water flows. Attenuation is removal over time and space; retention also refers to removal over time and space but is more commonly used in the European literature. Often, denitrification is not measured, but calculated by the difference in inflow and plant uptake/out flow.

2.3. NATURAL ATTENUATION PROCESSES

2.3.a Streams, rivers, lakes and ponds

We reviewed 26 articles deemed to be most useful on nitrogen attenuation in streams, lakes, and ponds. That work is synthesized in the discussion below. Boulton et al. (1998) describe the nature of water flow within the hyporheic zone of rivers and streams (the permeable region under a streambed, where water circulates in the interstitial spaces, and recharges or discharges into the groundwater zone or back into the stream flow), the potentially anoxic region of the bottom where denitrification can occur. The permeability of the bottom layer determines how fast water can flow through it, typically from 0.00001 m/s to 0.01 m/s, and the rate times the cross section determines the volume that flows through the hyporheic zone.

To determine how effective the stream can be for denitrification, the hyporheic flow volume must be compared with the flow through the volume of the stream itself where flows are typically from 0.1 – 2 m/s and the flow volumes vary from 5 to 10,000 times the flow through the hyporheic zone. Shallow, broad streams are more effective denitrifiers than deep streams because there is more hyporheic zone per volume of water passing through the system. In general, small, slow moving streams (<3m in width), as long as they have anoxic sediments, will be more effective denitrifiers than larger ones. Hill and Sanmugadas (1983) studying a small creek (<2 m in width) where there was less than 10% silt and clays in otherwise sandy sediments, found the highest rates of denitrification (250 mg NO_3-N m^{-2} day^{-1}) occurred in pools where decomposing algae accumulated on the surface. Swifter areas without algae had rates of less than 100 mg NO_3-N m^{-2} day^{-1}, emphasizing the importance of low flow rates and the availability of labile organic matter both for creating anoxia and for supplying food for denitrifiers.

Lakes and ponds are subject to similar restraints on denitrification rates. Detention times for the nitrate bearing waters are the critical factor; with enough time in a lake (two to three days) there is significant denitrification. In their excellent summary Jansson et al.

(1994a) state "The water retention time is the most critical single factor for removal of nitrogen. Thus, lakes remove more nitrogen than small wetlands even though the specific nitrogen retention … is generally considerably higher in wetlands." Saunders and Kalff (2001) summarize by stating that wetlands retain the highest proportion of nitrogen followed by lakes and distantly by rivers and that the differences are explained almost entirely by water retention times.

We summarize by saying that, in general, the larger (wider, deeper, higher flow) the stream, river, lake or pond, the less chance there is for dissolved nitrate to be retained in anoxic zones where denitrification can occur. These anoxic zones are found mostly in moderately porous muddy bottom sediments with adequate amounts of labile organic matter. The water carrying the N has to flow through or mix into these sediments at rates low enough (i.e. with detention times of hours/days rather than minutes) for oxygen to be consumed and denitrification to occur before reentering the water body. A deep pond or lake (>3 m in depth) will intercept more groundwater than a shallow pond. For a pond/lake deeper than ~2 m, groundwater should seep into the pond through the bottom slowly enough that channels free of fine particles are not created. Such channels would prevent denitrification from taking place. A shallower pond can function like a stream where the dissolved N mixes into the hyporheic zone.

A stream or pond must have a muddy bottom to function as a denitrifier (low oxygen, labile carbon), so currents must be slow enough to permit accumulation of muds. Sinuosity functions to lengthen the stream and increase the chance of denitrification by providing more time for the bacteria to work. Note that some management options such as dredging, while they may improve flushing, may diminish natural nitrogen attenuation. Temperature and therefore season and shading will affect denitrification rates since rates increase by about two fold with a 10° C temperature rise. Since groundwater temperatures in Massachusetts range from about 5-11° C some denitrification will occur throughout the year. A shallow (<2m deep) pond in a sunny location, with more than a day's detention time, and protected from the wind is likely to be a good denitrifier. A strong wind will mix oxygen into the system reducing its effectiveness as a denitrifier. Presence of other contaminants (assuming they do not damage organisms) or pathogens will have little effect on N processing.

2.3.b Natural bogs or fens

There is little information on the use of bogs or fens for N removal. We found 10 papers that mentioned nutrient cycling; please refer to Appendix A for the complete list and Appendix B for the annotated articles. These papers had little quantitative data and provided few insights into the denitrification process in bogs and fens. These systems generally have a peat substrate that is usually not very porous. N processing would then occur on the surface or in groundwater flow immediately beneath the peat.

2.3.c Wet meadows/Freshwater emergent/constructed wetlands

Approximately 60 papers fit into this category. Some papers related data and research on actual wetlands, some on constructed wetlands, some of the papers were reports on modeling and some were reviews of the work of others. Wet meadows, freshwater emergent and submerged wetlands are shallow systems often at locations where groundwater discharges; in theory, they should be good denitrifying systems. The important requirements to make these good denitrifying systems (similar to shallow ponds discussed above) are to have the dissolved N enter the anoxic regions of the wetland after nitrification has occurred, to have groundwater distributed as uniformly and shallowly as possible within the wetland or wetland substrate, and to have a detention time of one day or longer. If the water is >0.5 m deep, denitrification will also depend upon the slow process of diffusion to transfer nitrate from overlying water to anoxic zones.

Constructed systems are designed so that water flow is distributed uniformly through the system to avoid channelization of flow and to maximize contact time with anoxic soils. Most of the data (as can be seen from reviewing the titles in Appendix A) on N treatment in freshwater marshes are from systems constructed for wastewater treatment or from mesocosms (experimental systems large enough to approximate the essential parts of the nitrogen cycle in natural systems). Crumpton et al. (1993) working with mesocosms showed nitrate loss could be "described as a first order process (the rate is proportional to concentration) over a wide range of concentrations, and loss rates can be modeled based on factors controlling the rate of nitrate flux to anaerobic (anoxic) sites." (Parentheses inserted by authors). Their experiments used 3 to 15 mgl^{-1} nitrate nitrogen, residence times of about a week, and found 80 to nearly 100% nitrate loss. In a subsequent experiment using water from the Des Plaines River in which the nitrate concentration varied from over 6 mg l^{-1} in spring and fall to essentially zero in summer, they found 78-84% removal in wetlands with an average 12 day retention time and 95% removal in a wetland with a three month retention time (Phipps and Crumpton 1994).

It is important in looking at the review papers to be critical of the values presented. Howard-Williams (1985) lists denitrification rates in wetlands but does not indicate whether the reported rates are from the system as it functions or from an experiment or calculation that maximizes rates. For example, the two highest values in his Table 7 are both misleading if taken to be representative of what occurs in natural systems. The 550+ mg N m^{-2} day^{-1} is from mud shaken with water in a flask, and the <7000 mg N m^{-2} day^{-1} value is estimated from an assumed ratio of N_2O /N_2. In a later paper Howard-Williams calculated the rate of denitrification for the natural, undisturbed system to be 10 mg N m^{-2} day^{-1}.

Constructed wetlands for wastewater treatment can be any size, the water and sediment combined depth range to about 3 m, and organic content is low following construction but increases over time as the vegetation matures and cycles. Grain size ranges from coarse sand to rock/rubble to ensure subsurface flow. Constructed systems often use *Phragmites* or reed canary grass, invasive species (undesirable in natural systems) because they are hardy, spread rapidly, and provide a ready source of labile carbon. These invasive species can be replaced by the native sedge *Schoenplectus* that also performs well in these systems. To maximize denitrification in these highly managed systems, most are designed with detention time of 12 hours or more. Many of the constructed systems rely upon carbon addition to maintain high rates of denitrification at lower detention times. These systems are somewhat buffered from seasonal air temperature changes because they are groundwater mediated. Studies in Sweden indicated that alternating emergent (eg. cattails or reeds) and submerged vegetation (eg. pondweeds or milfoils) areas along the water flow path could enhance water distribution, reduce flow rates, and hence denitrification (Weisner et al. 1994). They found denitrification rates from 21 to 37 mg N m^{-2} hr^{-1} (1500-2640 µmol m^{-2} hr^{-1}) in experimental systems to which 1.4 g of nitrate-nitrogen per square meter had been added. In the few natural wetlands where there are data for latitude similar to Massachusetts, the rates are 0.01 to 0.1 mg N m^{-2} hr^{-1} (0.7-7 µmol m^{-2} hr^{-1}) (Johnston 1990).

2.3.d Cranberry bogs (in agricultural use or abandoned)

Only one paper dealt with cranberry bogs. Active cranberry bogs may denitrify, especially when flooded, but since they are fertilized they are net sources of N rather than sinks (Teal and Howes 1995). Diverting N bearing water through them could conceivably substitute for fertilizer additions but would have to be carefully monitored and managed to prevent over fertilization that would diminish the crop yield. This has been successfully completed at one location. While we do not have the data or a report on this project we have included the reference to the Willow Bend development in the Popponesset Bay watershed[2] where this experiment was carried out. This project has been successfully carried out and monitored.

Diverting groundwater through an abandoned bog built in a wetland (not the modern bogs built in upland), assuming there is labile organic material in the sediment and assuming detention time could be managed, would achieve a high rate of denitrification because this type of system comes closest to resembling constructed wetland systems used for denitrification. These systems have a decided advantage over natural bogs because most have water control structures in place to intercept surface water and groundwater that allows for managing the detention time. One caution concerning these systems is the potential for abandoned bogs to have retained nitrogen fertilizer. If pore water samples show high nitrate in an abandoned bog, flooding the area to create anoxic conditions should stimulate denitrification before the water is released to a holding pond or stream. Leonardson et al. (1994) report that alternate flooding and drying reduce nitrogen retention rates and that best performance in flooded meadows was from continual rather than intermittent inundation.

2.3.e Woody and open wetlands/riparian zones

Wetlands covered with woody vegetation such as swamps and floodplains have characteristics similar to those in the wetlands discussed above. Riparian forests also denitrify. We looked in detail at 45 reports for such systems. In all cases the surface or groundwater containing the dissolved N must enter the soils in the anoxic region and there must be enough labile carbon to support denitrification. Vought et al. (1994) found that nitrogen in surface runoff was removed linearly in proportion to distance from the source, 20% by 8 m and 50% by 16 m, but in subsurface flows about 80% was removed in the first 8 m and nearly 100% after 20 m. Unfortunately the paper does not give loading data, just percent removal.

[2] Personal communication with Massachusetts DEP, March 2007

The ideal width for a riparian zone depending on surface flow would be about 30m. Vought et al (1994) also found that the type of vegetation in the buffer strip in southern Sweden didn't make much difference, i.e. brush, pasture or forest worked almost equally well, as long as the area was undisturbed. However, studies in Illinois showed that forested riparian buffers were more effective than those covered with grasses.

Because of variability in the many factors influencing nitrogen removal in riparian zones, precise predictions on N removal rates are not easily made. Addy et al. (1999) "caution against ascribing specific ground water NO_3 –N removal rates to different riparian aboveground vegetation types without recognizing the importance of site differences, e.g. water table dynamics, land use legacy and adjacent vegetation. Riparian zones composed of a mix of forested and mowed vegetation, common in agroforestry and suburban land uses, may remove substantial amounts of ground water NO_3-N." In areas where the water table fluctuates widely, changing the depth of the oxic zone, denitrification would be less predictable than in those areas with saturated anoxic zones. The land use history would be a predictor for the amount of organic material available in the top three meters of soil. The adjacent vegetation types would give clues about carbon availability from surface litter and could be used as an indicator of how effective a buffer might be for removing nitrogen from the system.

In systems with only periodic flows it may take a few hours for the microbial denitrifiers to become fully activated. Significant denitrification has been found to a depth of 3m in a riparian wetland as long as the soils had an adequate source of labile carbon to that depth. In coarse soils like glacial outwash, denitrification declined abruptly with depth because there was less labile carbon and there was not an adequate anoxic zone to complete the denitrification process (Kellogg et al. 2005; Hill et al. 2004). Alder stands fix nitrogen even if groundwater inputs are relatively high in N so alders may not be good choices for woody treatment systems although they do stabilize stream banks effectively.

2.3.f Salt marshes, ponds and mud flats

Twenty-two papers on salt marshes were reviewed for this wetland type. These studies showed that salt marshes denitrify effectively with flows that enter marsh creeks, flow over the marsh surface, or flow through permeable layers immediately under the marsh sediments. If groundwater flow is too deep (more than about 1 m below the bottom of the peat; peat depths vary with each marsh and within a marsh), there may be insufficient organic matter and perhaps too high a redox potential for denitrification to occur as Nowicki et al. (1999) found in Nauset marsh. White and Howes (1994) found that of N added as NH_4^+, 40% was sequestered in marsh sediments and 54-77% of the remainder was denitrified after nitrification (equivalent to about 62 μmol N m^{-2} hr^{-1}), the rest exported. This study was conducted over a 7-year period and illustrates that in environments where there is a significant accumulation of organic matter (e.g. a salt marsh keeping up with sea level rise, or a forest accumulating wood) a substantial amount of nitrogen can be sequestered over a long time. Davis et al. (2004) found nitrogen flux into (N fixation) and out of (denitrification) Rhode Island salt marshes correlated with how much nitrogen was coming into them from land sources. Increased flux into the marsh decreased fixation and stimulated denitrification with the latter rates reaching 710 μmol N m^{-2} hr^{-1}. From water flowing under these marshes Addy et al. (2005) found that from 29 mg N l^{-1} during the cool season to 60 mg N l^{-1} during the warm season were denitrified, probably supported by organic matter coming from the overlying marsh.

Salt ponds are integral to salt marshes and their role in nutrient cycling has not been separated from marshes in general because of a lack of site specific papers and data. It seems unlikely that salt ponds intercept groundwater flow to a significant extent. Mud flats function in much the same way as salt marshes. Deep processes are the same year round; surface processes slow in winter because of the cold.

3.0 SITE MODIFICATIONS TO ENHANCE NATURAL ATTENUATION

The mechanisms contributing to nutrient retention include sedimentation, uptake and long-term storage in vegetation, and denitrification. Changing factors that limit nitrogen uptake or denitrification may enhance these processes. As discussed above, the most effective way to enhance nitrogen removal from surface or groundwater flowing through wetlands or waterbodies is to enhance denitrification. Denitrifying conditions are influenced by the hydrologic regime, organic carbon content, temperature, and substrate conditions including redox potential and pH. See Table 1.

3.1. DESIGN CRITERIA

Assuming the ground or surface water to be treated is nitrified (i.e. it is in the form of nitrate) and assuming that the nitrate is in the upper layer of groundwater such that it has not been diluted or submerged by down gradient infiltration, the following design criteria need to be considered:

Groundwater should:

Erupt on to the surface as a seep into a pond or wetland, or flow through a pond bottom, re-emerge into a stream that flows into another, downstream pond or wetland, or flow close to the soil surface (i.e. shallow groundwater), or be able to be brought near the surface to encounter a denitrifying environment, described below.

Surface water should encounter a denitrifying environment, described below, that can treat it before it enters a river, stream, coastal ocean or enters groundwater.

3.1.a Denitrifying Environment

The denitrifying environment is affected by a number of factors, including the following:

- Location of denitrification – Denitrification in riparian buffers may occur within 3 m of the ground surface, but is most efficient when it occurs within the top 1 m. Conditions vary from site to site with an abrupt decline in denitrification rates with depth in glacial outwash sediments and with little decline in alluvium (Kellogg et al. 2005). Denitrification occurs in streams in the hyporheic zone. Denitrification in ponds occurs in the bottom sediments. Denitrification occurs under fresh or salt marshes within 1 m of the base of the peat layer. The peat layer thickness depends on the age of the wetland and location. Redfield (1972) provides a good example of how peat depths vary within a Barnstable salt marsh.

- Nitrogen loading – ground and surface waters with higher nitrogen concentrations (NO_3 ~ > 3 mgl^{-1}) are more treatable than those with lower concentrations (e.g. Casey and Klaine 2001) because higher loading supports a healthier more stable denitrifying microbial population.

- Substrate – the substrate through which groundwater flows must be anoxic (have low redox potential) and be porous enough to allow flow (sand/gravel mixed with woody debris, some clay or silt). The substrate over which surface water flows can be finer (that is, with a greater proportion of silts and clays) as long as the water mixes into the hyporheic zone.

- Organic carbon concentration - labile organic carbon is needed to support the denitrifying microbial population. In natural systems, the source is generally decaying vegetation. In artificial systems, carbon is added to ensure denitrification: for each mg of nitrogen in NO_3, 2.7 mg of methanol are used, or, 1 mol nitrate requires 1.18 mol of methanol (Crites and Tchobanoglous 1998).

- Temperature -- denitrification rates approximately double for each 10º increase in temperature experienced by the microbes between 0º and 20ºC. Rates continue to increase with temperature, but microbial function is impaired at temperatures approaching 40ºC. If the temperature is near 0ºC, denitrification rates are less than half of what they are at 10ºC. Rates double from 10ºC to 20ºC (Crites and Tchobanoglous 1998). In Massachusetts near the coast, groundwater temperature is generally about 10ºC throughout the year while surface water temperatures vary seasonally.

- pH -- pH > 6 is required to support nitrification. The optimum pH for denitrification is 6.5 to 7.5 units. In Massachusetts, soil pH is typically below 6 from acid rain and other factors, so for optimal nitrogen treatment the addition of lime may be needed at rates determined by soil tests.

- Detention time – to enhance denitrification, nitrate-laden water should be in the anoxic zone, with labile carbon, at 10ºC for a minimum of 12 hours. Most of the natural systems discussed in the *Ambio* 1994 articles suggest up to three days detention in natural systems. If flows are intermittent, a few hours may be necessary to activate the denitrifying bacteria (Casey and Klaine 2001).

3.1.b Site Modifications

Site modifications can change the extent to which each of the wetland or waterbodies denitrify. Local site conditions (depth to groundwater, amount of nitrate in groundwater, sediment permeability, and the characteristics of a denitrifying environment listed above) will have to be determined and incorporated into design for system modifications if nitrate removal is to be effective. The modifications will be most effective ecologically and economically if they result in self-sustaining systems that do not need constant maintenance to function. Thus we do not recommend alterations that would require continuous carbon addition by humans.

3.1.b.i Streams, rivers

i) Slow flow (< .1m/s) is important for maximizing nitrogen processing. Slow flow can be promoted by creating meanders (lengthening the stream by reducing the hydrologic gradient), or restoring natural sinuosity or by encouraging natural obstructions (fallen logs, beaver dams, submerged aquatic vegetation) (Groffman et al. 2005). These actions encourage anoxic conditions at the water sediment interface, increase detention time, increase sediment deposition (carbon source) and enhance a hyporheic zone that is effective for denitrification. Stream flows for rivers gauged by USGS in southeastern Massachusetts are available at http://waterdata.usgs.gov/ma/nwis/current/?type=flow.

ii) Allow overbank flooding onto flood plain and/or riparian zone to increase contact with a denitrifying environment along riparian zones, and

iii) Increase available organic matter (fallen logs, beaver dams, submerged aquatic vegetation) to provide a source of labile organic matter that microbes can use in denitrification.

Direct or indirect adverse effects include: changes in fauna, increased water temperature, and an opportunity for non-native plants to invade. Other direct effects may be change in wetland type, altering wetland buffer/setback zones on newly created wetlands, and monitoring requirements for the wetland system.

3.1.b.ii Lakes and ponds

i) If there is a pond immediately down gradient of a nutrient source, modify it to deepen at the upgradient end (to capture more groundwater), leaving the down gradient end shallow for denitrification. Consider flow control structures to increase detention time (dams, berms, weirs).

ii) If there isn't a pond immediately down gradient of a nutrient source, create a shallow pond (depths ~ 2m) by excavating into groundwater where it is close to the surface; adding leaves and mud after excavation to enhance anoxia by providing carbon, although natural deposition over time of organic matter from algal growth (phytoplankton and benthic algae) will create the desired conditions.

iii) Create shallow embayments on lakes similar to shallow ponds above, and

iv) Do not remove muddy pond bottoms (if excavating, save and put back) to ensure an anoxic bottom layer with adequate organic content. If carefully done (e.g. using silt curtains) this should diminish water quality in the pond only temporarily.

Most ponds in coastal Massachusetts are connected with groundwater; thus the surface of the pond is indicative of the depth to groundwater under the adjacent upland. Ponds may also be "perched" on an impervious layer, i.e. vernal pools. These wetlands are fed by rain and do not indicate groundwater depth. The locations of vernal pools are generally known locally and are protected by the Wetlands Protection Act and thus should not be considered for modification.

Direct or indirect adverse effects include: short term effects from disturbance, the potential for eutrophication if overloaded, wetland filling/loss, attractive nuisance for waterfowl with possible pathogen and nutrient addition.

3.1.b.iii Natural bogs or fens

Preserve them until more is known about nitrogen retention in these systems.

3.1.b.iv Emergent and shrub-scrub wetlands

Divert nitrate rich water into these wetlands (and enlarge the system if needed). If the flow is channelized, reduce channelization (by regrading; distribution box; strategic use of woody debris). If there is space, an upgradient pond would be a positive addition; it would collect ground water, provide nitrification if needed and some denitrification. A down-gradient pond is not as useful as an upgradient one (Fleischer et al. 1994).

Direct or indirect adverse effects: There may be a loss of diversity in infertile wetlands with N additions, with the potential for rare species being lost (Moore et al. 1989).

3.1.b.v Wet meadows

Same management as for emergent and shrub-scrub wetlands

3.1.b.vi Cranberry bogs

i) In agricultural use -- Diverting groundwater with high N levels onto a working bog to substitute for fertilization of the bog would generally be difficult because of the level of monitoring necessary to adequately fertilize but avoid over fertilization that can cause increased susceptibility to disease and reduced yield (e.g. as described in www.tricity.wsu.edu/~cdaniels/profiles/cranberr.pdf)

ii) Abandoned – Evaluate water control structures to determine if they could be used to divert NO_3 rich water to the site, preferably by gravity. If the bog can be flooded, design the flood system for a detention time (> 12 hours) in the anoxic zone to ensure denitrification. The flow may be slightly subsurface or completely flooded (as a shallow pond); add carbon as needed. If the bog was originally a wetland but is now above groundwater level, remove the agricultural sand layer to restore hydrology.

Direct or indirect adverse effects: disposal of agricultural material; cost to manage; attractive nuisance for waterfowl

3.1.b.vii Forested and open wetlands

Allow over bank flooding onto a floodplain that has not been isolated from the stream or river regardless of what type of wetland it contains. Restore riparian wetlands on stream and river edges, ideally at least 30 meters in width. Mayer et al. 2006 reported probable 75% nitrogen removal efficiency in a buffer of 28 m. For surface flow, a buffer nearly 120 m wide was needed to achieve the same efficiency.

Direct or indirect adverse effects: Flooding of residential or commercial areas.

3.1.b.viii Freshwater emergent marsh wetlands

Same as for emergent and shrub-scrub wetlands.

3.1.b.ix Salt ponds

See salt marshes below.

3.1.b.x Salt marshes

Allow springs along the upland edge of the marsh to emerge onto the marsh surface. Allow marshes to move inland as relative sea level rises. Preserve existing marshes, restore marshes where flows have been disrupted, and create marshes where they have been destroyed. It is important that salt marshes be freely connected to tidal exchange so that they maintain pH and anoxic sediments. If the salt marshes are isolated by culverts or small bridges they are subject to conditions that can make salt marshes highly acidic. These conditions will result in killing of vegetation and other organisms, and reduce their treatment value severely. The marsh design should ensure that local groundwater flows will enter the marsh rather than flow under it. A salt marsh downstream from an active cranberry bog could be very useful for treating bog runoff if that runoff was diverted into the marsh. Most of the denitrification takes place in the bottoms of tidal creeks where the permeability is typically much higher than in the marsh peat (Howes et al. 1996).

We recommend that diversions of nitrate containing ground water be made into the upper ends of salt marsh tidal creeks rather than onto the marsh surface, although if springs are already emerging onto the marsh surface, don't interfere with them.

Direct or indirect adverse effects:

Freshwater flows into, onto and under marshes as ground and surface water and rain falls upon them. Most freshwater additions are small in proportion to tidal flows and so there is usually little change in salinity within the salt marsh and therefore little effect on marsh plants or organisms. Torrential rains can have episodic adverse effects. If enough freshwater is diverted onto the upper edges of a salt marsh plain so that salinities on the marsh plain are reduced (by 2 to 3 ppt), there is the possibility of encouraging invasive form of *Phragmites australis,* which thrives at salinities below about 20 ppt. (Silliman and Bertness 2004). There are correlations with *Phragmites* invasion of upper borders of salt marshes and housing density that Silliman and Bertness (2004) attribute to increases in both nitrogen and freshwater inflow. However, in Eastern Europe high nitrogen levels (e.g., greater than 0.2 mg l^{-1} NO_3 and 8 mg l^{-1} total N), even in freshwater systems where salinity stress is absent, are considered to be a cause of *Phragmites* decline (Santrueckova et al. 2001).

Nitrogen additions made to the marsh surface can alter the balance among species of higher plants on a salt marsh, generally favoring tall *Spartina alterniflora* (Emery et al. 2001, Wigand et al. 2003). Again, we recommend introducing additional nitrate laden water into the heads of tidal creeks. Note that even with more *Spartina alterniflora* on the marsh plain, this would only make a New England salt marsh more like a mid-Atlantic marsh. This could reduce habitat for some high marsh species such as harriers and increase nursery habitat for marsh fishes.

Nitrogen additions in themselves can change the productivity and diversity of a salt marsh vegetation and infauna, but even at very high levels do not alter salt marsh functions (Valiela et al. 1982).

3.1.b.xi Mud flats

See salt marshes above.

3.1.b.xii Other Modifications

i) Reactive barriers or organic filters that intercept groundwater flow. Excavate a ditch perpendicular to ground water flow and install a barrier/filter such as hay bales (to be a carbon source and attachment site for bacteria).

Direct or indirect adverse effects include: the cost to install and maintain pumps, piping, filters; filter or barrier replacement/disposal.

ii) Locate wastewater treatment plant effluent to take advantage of wetland or salt marsh polishing by piping nitrified discharge to a manifold for distribution to soil in riparian forest, wetland, or head of a salt marsh creek.

Direct or indirect adverse effects: interbasin transfer of water, change in groundwater table, potential pathogen introduction if not disinfected, potential change in salinity and/or salt marsh vegetation (species distribution and biomass).

3.1.c Avoiding adverse effects on existing wetlands and waterbodies

1) Manage flow so that detention time is adequate for denitrification.
2) Manage flow so storm events are not likely to wash away the anoxic organic sediments needed for denitrification (diversion pathways; over bank flooding, etc.) or wash away the as-yet untreated nitrate.
3) Do not load a pond or lake to the point of complete eutrophication. These systems are typically phosphorus limited rather than nitrogen limited and phosphate is usually not carried far by groundwater. See Section 5.2 and 5.4.
4) Monitor the sites, perhaps using citizen monitoring groups as described in the DEP manual, online at http://www.mass.gov/dep/public/volmonit.htm#guidebook

3.2. CARRYING CAPACITY

Carrying capacity can have two meanings. First, it can mean the amount of nitrogen that can be diverted into a water body without altering its biological and chemical functions. Second, it can mean the amount of nitrogen that will saturate the wetland's ability to denitrify, with the excess passing through the system without being treated.

The carrying capacity of wetlands varies widely depending on wetland characteristics, especially conditions that influence nitrogen storage and denitrification rates. Carrying capacity depends on nitrogen loading rates, with higher nitrogen retention (and denitrification) in wetlands with higher nitrate loading rates (up until carrying capacity is met, at which time denitrification rates increase more slowly or not at all). A system with low loadings may develop higher treatment capabilities with increased loadings. A goal of natural attenuation projects should be to add nitrogen as nitrate and to denitrify it as efficiently as possible such that little of it is taken up by plants, stored in sediment or lost to the outflow.

Diverting NO_3-rich ground or surface water could potentially lead to nitrogen overload (exceeding the carrying capacity). In the case of ponds and lakes, this can lead to eutrophication, algal overgrowth, fish kills, etc. Phosphorus is generally acknowledged to be the limiting factor for primary production in freshwater systems, although not always. Even if the addition of nitrogen did not cause eutrophication immediately, it could occur over time. For example, if septic systems near a pond are the source of nitrate, and the subsoils are relatively clean sands and gravels, their ability to retain phosphorus will eventually be exhausted. Transport of phosphorus in ground water can then increase with nitrogen eventually becoming the limiting factor in plant growth (Crites and Tchobanoglous 1998).

<u>High loading to salt water systems</u> Howes et al. (1996) showed that when 7.5 g N m^{-2} wk^{-1} were applied to experimental sites from early May through mid November every year since 1974 at Great Sippewissett salt marsh, 2/3 was denitrified. This addition rate was chosen by John Teal in 1974 when the experimental work on Sippewissett salt marsh started because it was over 10 times that recommended for the most demanding pasture grasses in Massachusetts. These experimental plots are still healthy after 30 years of fertilization at this rate (Teal, 2006, personal observation). Repeat of statement on prior page.

<u>High loading to fresh water systems</u> Peterson and Teal (1996) showed that when 5.23 g N m^{-2} d^{-1} were applied to fresh water constructed wetlands, 37% was denitrified in about one day detention time. At half the loading and twice the detention time, >50% was denitrified. The vegetation was not adversely affected by this loading. On the other hand, Moore et al. (1989) found higher species richness and more rare species in infertile marshes, those with low standing crops (< ~400g m^{-2}), than in fertile ones. So diversion of nutrient rich waters into an infertile wetland could have undesirable effects, which would offset the desirable result of reducing coastal pollution. It is highly unlikely that NO_3 in ground or surface water will approach levels near 5 g N m^{-2} d^{-1}.

Methods to determine the maximum allowable new NO_3 that can be introduced to a wetland or waterbody.

a. For complete denitrification, and
b. Without altering the waterbody/wetland.

i) Experiment by spiking the influent with increasing amounts of NO_3 and measuring the outflow. When NO_3 rises in the outflow (e.g. in a marsh creek at low tide), the maximum loading has been obtained.

ii) Manage the diversion of ground or surface water into the wetland or water body and measure the outflow (e.g. an exit stream). Stop adding new flow when NO_3 in effluent rises. It may require 12 to 24 hours depending on calculated detention time in the anoxic zone for the system to respond.

It would be best to rely on physical measurements; it is very difficult to accurately model this function because of the complex interactions within the ecosystems. These measurements are commonly made in constructed ponds and wetlands for wastewater treatment.

Wetlands, as mentioned above, generally can deal with much more nitrogen than lakes or ponds. There may be changes in the vegetation: annual height, growth rates, species mix and other effects. Changes in vegetation may increase, decrease or alter the way an area is used by insects, small mammals and birds. For example, more nitrogen in marsh plants can make them more attractive to herbivores so grazing may increase. There may be changes in infauna as well. There will need to be a discussion of these tradeoffs in the public process for permitting natural attenuation projects. One question that will need to be answered is: what changes are acceptable to reduce N loading to the coastal environment? Neither the data reviewed here nor our abilities to model complex ecosystems are adequate to predict the exact changes that may occur to a wetland or waterbody from increasing flow and NO_3. If more N is added to the system and the rate of denitrification does not increase, then the system will become more eutrophic. In most cases, the actions are reversible provided the wetland system has not been physically altered and there are options for increasing denitrification rates and or decreasing N loads.

In all of the systems the real danger from overloading would be the failure to denitrify sufficiently to protect the down gradient systems from eutrophication. In other words, if a treatment system is so highly loaded that the nitrogen passes through unchanged, the whole purpose of the project is lost.

3.3. FROM IMPLEMENTATION TO PREDICTED RATES OF NITROGEN ATTENUATION.

New ponds or dredged ponds may not immediately have the appropriate anoxic sediment types needed to support natural nitrogen attenuation unless organic materials are provided during construction. Sedimentation rates will vary according to watershed size, upstream and pond bank erosion, and other factors and are important for supplying labile organic matter. Leaf litter additions and macrophyte contributions may take a growing season before they became a positive source of carbon.

How should these factors be incorporated into estimates of enhanced nitrogen attenuation for a particular project? The microbial populations develop rapidly (hours to days) and begin to denitrify once conditions are suitable (anoxic and with carbon supply). The development of an environment for denitrification could take years if a constructed system started without an organic substrate or carbon additions. Mitsch et al. (2005) found hydric soils developed in two to three years with an increase of 1% per three years in organic content of the substrate. If the system is constructed with hydric soils then denitrification could begin within days or as soon as the construction disturbance settles down.

To summarize, if the project is a simple nitrate rich water diversion into an existing pond or wetland that has all the denitrifying qualities listed above, one could expect an immediate increase in denitrification rates. If the project is new pond and wetland construction, full function may take from one to 3 years.

3.4. MONITORING AND ADAPTIVE MANAGEMENT

The critical property to monitor in wetlands used for nitrogen attenuation is the difference in fixed nitrogen (nitrate and ammonia) concentration between inflow and outflow of the system. Many of the system properties that control how well the system functions can be monitored quite simply: look to see that the water flows do not become so channelized as to prevent contact with anoxic parts of the system; observed substrate color and smell to see that it becomes and remains anoxic; observe in- and outflows to see that water has not been diverted into another system.

Monthly observations during Year One and quarterly samples thereafter should be adequate. Inflow and outflow can be measured in the field with an oxygen probe for DO and with Hach kits for ammonia and nitrate.

Since the purpose of these modifications is to reduce nitrogen transport to coastal systems, the critical monitoring is contained in the previous paragraphs. It is less valuable to monitor the receiving body unless an attenuation project intercepted a substantial fraction of ground or surface water flow. Until there are a number of projects, where their cumulative impact may be measured, it is likely that observations of benthic algal density and water transparency for indications of phytoplankton production will suffice since the rest of the ecosystem will respond favorably if eutrophication is reduced or prevented. Monitoring an attenuation project for habitat and wildlife species changes is desirable; for wetlands, the simplest and most efficient observations would be on bird use. Vegetation monitoring may also be useful with the understanding the vegetative change may be a positive outcome. For example, if one were modifying a cranberry bog, one would expect to see an increase in vegetative diversity. Whether or not this type of monitoring is required is likely to depend upon the scope, complexity, budget and magnitude of proposed N removal project.

Adaptive Management: If nitrate in the outflows is high (>50% of inflow value) and remains high over several sampling periods, one could either stop adding nitrate rich water so that the system would revert to its former state or determine which of the denitrifying parameters was not met and fix it.

3.5. SUMMARY OF POTENTIAL NITROGEN REMOVAL IN VARIOUS WETLAND TYPES

Although it is impossible to precisely predict nitrogen retention and denitrification rates in wetlands, a number of general principles apply. The most important factors promoting nitrate removal include a relatively high nitrate loading rate, long retention time to allow contact with denitrifiers, a supply of labile organic matter, anoxic conditions, relatively warm temperature, and neutral to slightly acidic pH. Under these conditions denitrification rates, and therefore nitrogen retention/removal rates, are optimal.

Table 1 shows conditions associated with high, medium, and low nitrate removal from various wetland types. Note that nitrate loading rates and water retention times are suggested in the table, but may not be common in natural systems. For example, streams with flow rates less than 0.1m/s are not commonly found in Massachusetts so nitrate removal would probably be only medium in local small streams. As another example, it is likely that suitable wetlands immediately down gradient of unsewered housing developments remove most of the nitrate from the groundwater that enters them; however, it is important to note that not all groundwater stays in the top two to three meters of soil where it is likely to be intercepted by the adjacent wetland. Some of the nitrate from the housing development will pass under the pond/wetland system with no opportunity for natural attenuation unless or until the groundwater emerges near the surface at the coast.

Table 1. Nitrate Removal by Natural Wetland Type

Wetland type	NO3 removal		
	High	Medium	Low
Lakes/ponds	anoxic sediments, permeable sediments, labile organic matter > 15%, >1 day residence time, pH 6.5 to 7.5, NO3 > 3 mg/l, > 10C	anoxic sediments, permeable sediments, labile organic matter ~ 15%, 1 day residence time, pH > 6, NO3 > 3 mg/l, > 10C	anoxic sediments, permeable sediments, labile organic matter < 15%, 12 hours residence time, pH > 6, NO3 ~ 3 mg/l, < 10C
Rivers/streams	anoxic sediments, permeable sediments, labile organic matter > 15%, >1 day residence time, pH 6.5 to 7.5, NO3 > 3 mg/l, flow <0.1m/s, > 10 C	anoxic sediments, permeable sediments, labile organic matter ~ 15%, <1 day residence time, pH > 6, NO3 ~ 3 mg/l, flow <0.1m/s, < 10 C	little hyporheic zone, flow > 0.1m/s, pH ~ 6, NO3< 3 mg/l, low organic matter, < 10C
Bogs/fens	leave as is till more understanding is available		
Wet meadows, emergent wetlands, constructed wetlands	subsurface flow, anoxic condtions, labile organic matter > 15%, pH 6.5 to 7.5, NO3 > 3 mg/l, > 10 C, >12 hr retention time, alterrnate submergent and emergent vegetation, relatively steady flow	subsurface flow, anoxic condtions, labile organic matter ~ 15%, pH > 6, NO3 ~ 3 mg/l, < 10 C, ~ 12 hr retention time, alterrnate submergent and emergent vegetation, variable flow or most flow in winter	subsurface flow, anoxic condtions, labile organic matter < 15%, pH ~ 6, NO3 ~ 3 mg/l, < 10 C, < 12 hr retention time, alterrnate submergent and emergent vegetation, intermitent flow
Cranberry bogs, abandoned	modify to become emergent wetland,		
Cranberry bogs, in use	don't use		
Riparian zones	> 30m width, subsurface flow <1m deep, labile organic matter > 15%, pH 6.5 to 7.5, > 10C	< 30m width but greater than 20 m, subsurface flow <2m deep, labile organic matter ~ 15%, pH > 6, ~ 10C	<30 m wide, deep subsurface flow, little labile organic matter in sediment
Salt marshes/ponds	if flow stays subsurface it should be within 1 m of bottom of peat, labile organic matter > 15%, tidal exchange to stay pH neutral and salinity above 20 ppt; ideally flow emerges into tidal creeks near land	groundwater >1m under the peat and may not emerge in tidal creeks; labile organic matter ~15%, pH >6, ~ 10C,	groundwater > 1 m under the peat and does not emerge in tidal creeks; little labile organic matter in sediment

4.0 DATA NEEDS, DATA GAPS, AND OTHER PROGRAMS

4.1. DATA NEEDED TO REVIEW AND PERMIT NATURAL ATTENUATION PROJECTS

4.1.a Watershed wide

Before considering a natural attenuation project, the following information should be assembled by DEP:

- A consensus on nitrogen sources: % airborne, stormwater, groundwater, point source
- A map of nitrogen concentrations in groundwater and surface water (measured and estimated)
- A map of water flow (volumes, travel times, etc.)
- A map of wetlands and waterbodies showing ownership (public, private, protected, etc.)
- A map of rare and endangered; unique habitat, etc.
- A map of the wetland plant distribution downstream of the proposed project
- Estimates of the amount of nitrogen that could potentially be removed.
- Estimates on costs; cost per unit nitrogen removed should be estimated.
- Evaluation of whether the project would produce a net benefit, with potential changes to the existing ecosystem understood as fully as possible.

- Assessment of whether the project could be permitted under existing laws and regulations.

4.1.b Wetland specific

a. The state of the literature and research on natural bogs and fens is not sufficient to develop techniques to use these wetlands as natural nitrogen attenuators without endangering either the resource itself or the downstream resources. We found that the data required to make modifications to natural bogs and fens are inadequate; we recommend they not be used for natural attenuation projects at this time.

b. For all other water bodies and wetlands (streams, rivers, lakes and ponds; wet meadows/freshwater emergent/constructed wetlands; cranberry bogs; woody and open wetlands/riparian zones; salt marshes, ponds and mud flats), the following data are needed to review and permit natural attenuation projects:

- Nitrate Concentration: Measure in situ nitrogen up and down gradient to confirm that the proposed site is not already over burdened with N. If the site is acceptable, measure nitrate concentration in ground or surface water proposed for diversion into the site. Try to determine the source of nitrate and estimate whether it is consistent, seasonal or event driven. Higher influent nitrate concentrations are preferred because the microbial biomass thrives at higher concentrations, leading to better denitrification rates. Also, from a cost/benefit perspective, the outcome (impact on the receiving water) is likely to be measurable if you choose to treat higher concentrations. If nitrate concentrations are below 3 mgl^{-1}, longer detention times are needed. Moreover, if ground and/or surface flows to the wetland or waterbody have periods with no nitrate, then several hours additional detention time at ~10°C is necessary to reactivate denitrification.

- pH: near neutral is best -- 6.5 to 7.5 units

- Anoxic conditions: required. This can be measured with a redox electrode.

- Temperature: best denitrification occurs when water temperature is 10°C or above. Some denitrification occurs at temperatures > 4°C.

- Detention time: Detention time in sediment depends upon the nature of the substrate (permeability) and can be estimated from soil samples. A minimum of 12 hours detention time in the anoxic zone of the wetland are required and two to three days detention time is preferred for ponds and lakes (Fleischer et al. 1994; Ahlgren et al. 1994).

- Labile organics: The nature of the substrate, the detention time in the anoxic substrate and the labile organic content affect the rate of denitrification. The labile organic content can be estimated as chemical oxygen demand (COD), which needs to be more than four times the concentration of the nitrate to be denitrified. Of course, this must be continuously replenished from local sources (decomposition or release from living organisms) for denitrification to continue. For example, sediment with over 15% organic content will probably suffice as will active growth of emergent and submerged herbaceous marsh plants.

4.2. DATA GAPS

The ability of a system, especially a newly created one, to supply sufficient labile organic matter to keep the sediments anoxic and supply the metabolic needs of the denitrifiers, is likely to be uncertain. Requiring the sediment in a created marsh to be a mix of coarse material and organic matter would provide some protection.

Uncertainties about detention times and denitrification rates can be resolved by field tests using tracers for flows and nitrate additions to test denitrification rates. This is particularly true for streams. Under the best stream conditions less than 0.1% of the flow goes through the hyporheic zone. But since the denitrification depends on flow in the hyporheic zone, exchange with the overlying surface flows, and the effectiveness of the hyporheic zone as a denitrifier, any calculation of denitrification rate is going to be extremely uncertain. So we believe for a stream to be considered, an actual field measurement of denitrification rate would be necessary. This would involve adding a known amount of nitrate (at a known rate) from upstream and measuring the NO_3^- concentration at both ends of the stream section under consideration. If the system did not perform as desired, it would be necessary to determine whether or not it was a lack of organic matter, permeability of the hyporheic zone, or too rapid flow that was responsible and the to make appropriate adjustments.

The likelihood of high rates of denitrification is best for wetlands and shallow ponds. There is considerable experience using aquatic systems and wetlands for wastewater treatment; the data are summarized in standard texts such as Reed et al. 1995, Kadlec & Knight 1996, or Crites and Tchobanoglous 1998.

4.3. POTENTIAL SITES FOR PILOT PROJECTS.

When siting pilot projects, it is important to consider basic information about site characteristics. The decision tree in figure 3 can guide the initial site selection process. Initial evaluation includes assessment of hydrologic and physical conditions at the site. Following this initial consideration, it is important to evaluate conditions at the candidate wetland site. Table 1 showed the important factors that promote maximum denitrification in a given wetland, including long retention time, anoxic sediments, pH 6.5-7.5, available labile organic matter, adequate temperatures and moderately high nitrate loading. If these are all present or can be easily created then the site is a good candidate for a nitrogen remediation site.

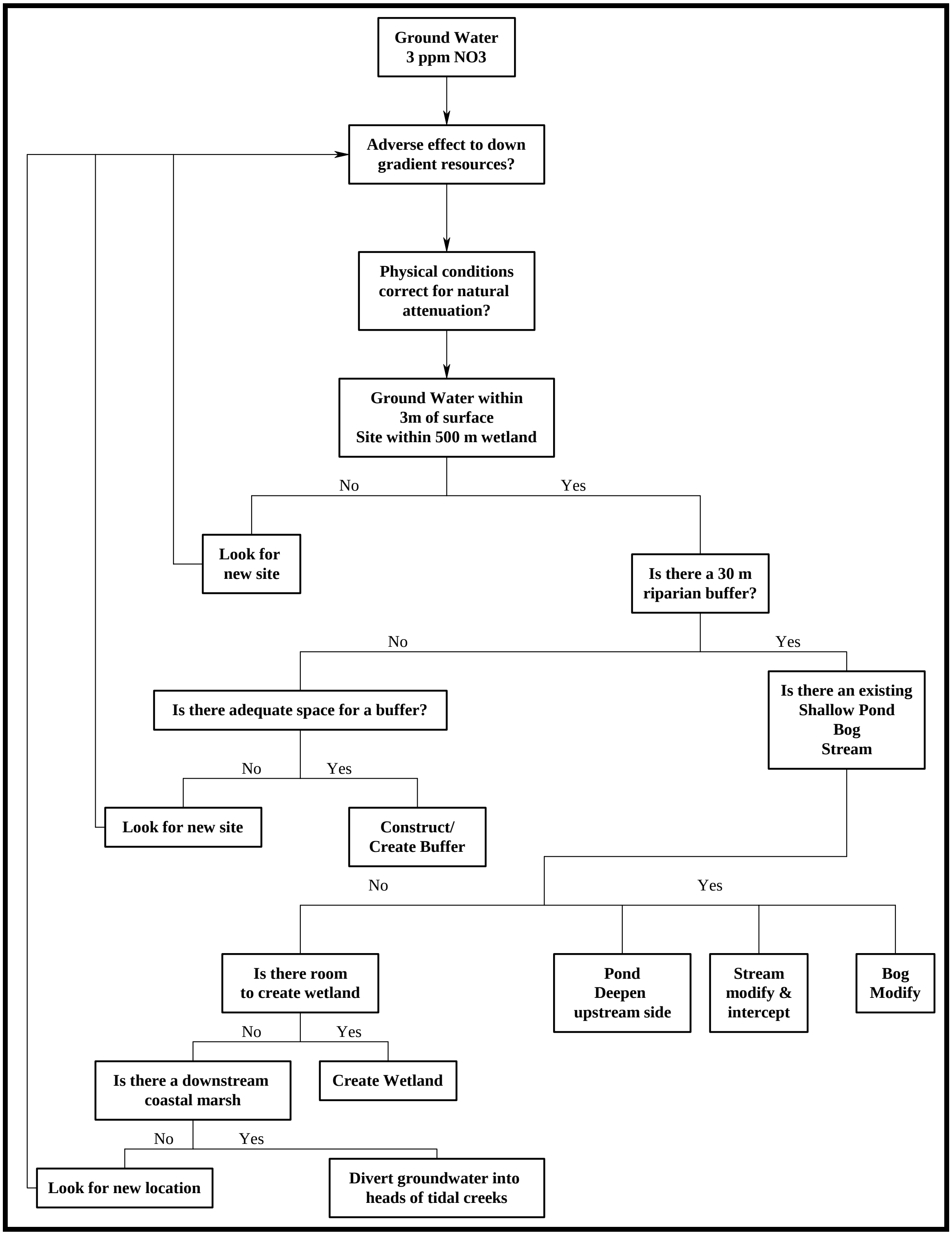

Figure 3. Decision Tree for N Attenuation Site Selection

Specific locations in Massachusetts where nitrogen retention projects could be pursued include the following:

Chatham: Muddy Creek area where there is some undeveloped land to create or enhance wetland in the upper watershed and a need for 3000 kg/year N reduction in the upper watershed to reduce loading to the embayment by ~40%.

Mashpee: Hamblin and Jehu ponds if the salt marshes around the edges of the ponds could be expanded.

Falmouth: Coonamessett River has a relatively high nitrogen concentration and is a significant contribution to the coastal ponds. Upgradient abandoned (or to be abandoned) cranberry bogs could be modified to receive river flow for denitrification.

Barnstable: Skunknett River had three shallow managed ponds (dating back to the 1890s) associated with it until control structures were destroyed in 1991 storms. Presently, the Skunknett River flows with a short residence time that is not conducive to nitrogen attenuation. Restoration of the ponds, if permittable, may enhance attenuation.

Mashpee: Santuit Pond in the Popponesset Bay watershed has a system of abandoned cranberry bogs that may provide some potential for attenuation. Control structures remain in place that could help manage flow and detention times.

Barnstable: Mill Pond in the Three Bays Watershed may provide potential as a demonstration for pond management (dredging, etc.) to gather data on attenuation and design factors to enhance attenuation in ponds.

4.4. PROGRAMS TO SUPPORT PROJECTS TO ENHANCE NATURAL ATTENUATION OF NITROGEN

In Massachusetts there is an ongoing natural attenuation project “Bioretention system to control nonpoint nitrogen pollution” (Suku Sengupta, University of Massachusetts, Dartmouth), funded by the Cooperative Institute for Coastal and Estuarine Environmental Technology (CICEET). Outside of the state there do not appear to be comprehensive, statewide programs. There are some regional programs within states, some very strong statewide stormwater programs that use natural systems, and a few federal programs that consider the idea.

New Hampshire: The UNH Stormwater Center has been evaluating the ability of constructed wetlands and bioretention systems to reduce nitrates and other pollutants. See www.unh.edu/erg/cstev/ for more information.

North Carolina: Urban Water Consortium has developed a program of Wet Detention Ponds for Phosphorus and Nitrogen Removal. The Urban Water Consortium is administered by the Water Resource Research Institute. Members include the Cities of Charlotte, Durham, Fayetteville, Greensboro, High Point, Raleigh, Wilmington, and Winston-Salem. http://www2.ncsu.edu:8010/ncsu/CIL/WRRI/

Contact Information: Dr. David H. Moreau, 919-513-1144 david_moreau@ncsu.edu

Maryland: Prince George's County has incorporated **Bioretention** as one of the main components of their Low Impact Development (LID) approach for managing stormwater runoff. "Bioretention was developed by Prince George's County around 1990 and has become the "Integrated Management Practice" (or IMP) of choice around the world. The County continues to refine the bioretention IMP design and its effectiveness has been documented by studies that show significant pollutant loading reductions." http://www.goprincegeorgescounty.com/Government/AgencyIndex/DER/ESD/Bioretention/bioretention.asp?nivel=foldmenu(7)

Contact information: Department of Environmental Resources (DER), Donna M.P. Wilson, Director. 9400 Peppercorn Place, suite 500, Largo, MD 20774 301-883-5810. DERcares@co.pg.md.us

NY and CT: completed an assessment of non-treatment alternatives for Long Island Sound. (A Total Maximum Daily Load Analysis to Achieve Water Quality Standards for Dissolved Oxygen in Long Island Sound. Prepared in Conformance with Section 303(d) of the Clean Water Act and the Long Island Sound Study (December 2000). That report summarizes their efforts as of late 2003: "The phased implementation of this TMDL continues the principle of *adaptive management* adopted by LISS. Considerable progress in understanding causes and consequences of hypoxia has been made over the last decade; similarly, much more is known about nitrogen removal technologies at sewage treatment plants and BMPs for the control of a wide range of nonpoint source and stormwater pollutants, including nitrogen. Management efforts in Connecticut and New York have benefited from adaptive management techniques. As they have become more progressive, the implementation phases have built upon the higher level of scientific certainty that the research and pilot studies have provided.

Adaptive management is especially critical to the success of achieving this TMDL because attaining water quality standards would require nitrogen reduction beyond the limits of current technology. It is expected that, over time, treatment technology will continue to improve, amendments to national and state air and water legislation will be enacted, and new and innovative funding sources will become available. The states of New York and Connecticut are prepared to respond to these changes. Similarly, EPA should begin to take more aggressive steps to control out-of-state air and water sources of nitrogen, in keeping with the level of understanding of their relationship to hypoxia in Long Island Sound.

Contact information: New York State Department of Environmental Conservation, 50 Wolf Road, Albany, NY 12233-0001 (518) 457-5400

Connecticut Department of Environmental Protection, 79 Elm Street, Hartford, CT 06106-5127 (860) 424-3020

California: San Diego Creek Watershed Project. Orange County, cities of Irvine, Lake Forest, Orange, Newport Beach and Tustin.

A discussion of the San Diego Creek Watershed Project notes: "The Natural Treatment System is a cost effective, environmentally sound alternative for handling dry weather runoff. In very simple terms it will work much like the San Joaquin Marsh with smaller manmade wetlands placed strategically in many areas throughout the watershed. Low-flow natural and urban runoff, as well as smaller storm flows will be diverted into man-made wetlands throughout the San Diego Creek Watershed where contaminants will be removed and prevented from reaching the Upper Newport Bay. At the same time the system will provide a natural resource, riparian habitat, wildlife and water quality benefits throughout the watershed."

Contact Information: Norris Brandt, P.E., NTS Program Manager (949) 453-5300; Irvine Ranch Water District P.O. Box 57000 Irvine, CA 92619 e-mail: brandt@irwd.com

Ohio

Ohio's Division of Surface Water has a great website and extensive reports on wetland mitigation banking, but we do not find any links to using natural systems for nitrogen attenuation.

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http://www.epa.state.oh.us/dsw/wetlands/WetlandEcologySection.html

http://www.epa.state.oh.us/dsw/wetlands/WetlandEcologySection_reports.html

Within the division of natural resources, there is a thorough manual on non-point pollution that promotes natural systems such a bioretention ponds and wetlands: http://www.dnr.ohio.gov/soilandwater/Coastalnonpointprogram.htm

Michigan

The Rouge River National Wet Weather Demonstration Project has developed wetlands for stormwater management: http://www.rougeriver.com/wetlands/index.html. This group has also inventoried their natural systems. We found no links to using natural systems for enhanced N removal.

More detail on stormwater and other wetland initiatives can be found in the nationwide references below.

Nationwide:

1) The Wetland Initiative

Wetland Inititiative reports note the following:

"Currently, state EPAs are mandated by the federal EPA to write and enact water quality standards for nitrogen and phosphorus or to adapt the proposed federal criteria. In response, Illinois water reclamation districts predict it will cost their ratepayers more than $5 billion to install the best technology available and $500 million annually to operate-yet even this investment will not produce water that meets the proposed federal criteria. Nor will it address a region's needs for open space, wildlife habitat, and flood control

The Wetlands Initiative and its partners[3] have developed an alternative: Restored wetlands, enhanced by the purchase of nutrient removal credits - either through an open market or through long-term contracts. We call this strategy "nutrient farming." In addition to removing nutrients, the restored wetlands will provide important environmental benefits for humans and wildlife-benefits that are unavailable through the use of traditional treatment technologies. The potential market for nutrient credits is huge. For example, an immediate market could be industrial and municipal dischargers.

Annual operating and maintenance costs at upgraded treatment plants are estimated to be $4.8 billion nationwide (extrapolated from Illinois per capita costs). This money would be better spent re-creating wetlands, which provide many other environmental benefits. Nutrient farming could be conducted throughout the Mississippi River Basin and other watersheds across the United States where high nutrient concentrations are of concern. The U.S. and Illinois EPAs have endorsed nutrient farming as a means to reach clean water goals.

Several major institutions are participating in far-reaching research to answer these and other questions. This interdisciplinary research will integrate knowledge of chemical, physical, and biological processes with the economic mechanisms necessary to support these landscapes. Although previous research established that wetlands reduce nutrients in surface waters under certain conditions and for certain time periods, scientists do not fully understand the complex interactions that produce this reduction or how to manage these processes. This research program will create a body of knowledge useful for predicting wetland reactions and outcomes over a wide range of conditions. With this fuller scientific understanding of wetland optimization, economists can evaluate the economic feasibility of nutrient farming over that same range of wetland conditions. Nutrient farming must be viewed as a business enterprise-an economically efficient means to manage environmental problems. This economic efficiency will be established only by using solid biogeochemical and economic data."

[3] The Wetlands Initiative, coordinator; Argonne National Laboratory; Bloomington-Normal Water Reclamation District; Bradley University; Greater Peoria Sanitary District; Iowa State University; Louisiana State University; Metropolitan Water Reclamation District of Greater Chicago; The Nature Conservancy; Northwestern University; The Ohio State University; Peoria Audubon Society; Sanitary District of Decatur; Springfield Metro Sanitary District; University of Illinois ; University of Minnesota ; University of Missouri-Columbia; University of Wisconsin

Contact Information: The Wetlands Initiative 53 W. Jackson Blvd., Suite 1015, Chicago, IL 60604 • (312) 922-0777, ext. 113 • Fax (312) 922-1823 dhey@wetlands-initiative.org • www.wetlands-initiative.org

2) Center for Watershed Protection

"Founded in 1992, the **Center for Watershed Protection** is a non-profit 501(c)3 corporation that provides local governments, activists, and watershed organizations around the country with the technical tools for protecting some of the nation's most precious natural resources: our streams, lakes and rivers. The Center has developed and disseminated a multi-disciplinary strategy to watershed protection that encompasses watershed planning, watershed restoration, stormwater management, watershed research, better site design, education and outreach, and watershed training."

Contact Information: CWP, **8390 Main Street, 2nd Floor, Ellicott City, MD 21043** Phone: 410-461-8323 Fax: 410-461-8324 email: center@cwp.org

Some examples of CWP's recent work are excerpted below. MA DEP is familiar with their work on the lower Charles River.

New York BMP Manual In 2001, the Center worked with the New York State Department of Environmental Conservation (DEC) to produce the New York State Stormwater Management Design Manual (the Manual). The Manual will be a key tool for DEC in implementing NPDES Phase II stormwater regulations statewide. As part of this project, the Center is training DEC regional staff on the content and application of the manual in order to implement it statewide. In addition, the Center will assist the DEC to update their 1992 publication entitled "Reducing the Impacts of Stormwater Runoff from New Development," which outlined the impacts associated with urban and suburban development, and strategies to reduce them. The update will incorporate new research on stormwater impacts, as well advances in the stormwater field that have occurred over the last 10 years.

Staff Contact: T. Brown

Builders for the Bay In partnership with the National Association of Homebuilders and the Alliance for the Chesapeake Bay, the Builders for the Bay is designed to bring together builders, environmental groups, local governments, and other stakeholders to begin the local roundtable process in communities around the Chesapeake Bay area. This project requires training the staff of the Alliance for Chesapeake Bay and selecting communities in which to conduct local roundtables throughout the Chesapeake Bay region. In 2003, the Center plans to continue working on Harford County's roundtable. Potential BFB roundtables for 2003 include Paxton Township, PA; Prince William County, VA; Anne Arundel County, MD; Worcester County, MD and James City County, VA. The Center also anticipates aftercare projects for Cecil and Frederick County, MD.

Staff Contact: H. Kwon

Vermont Stormwater Management Handbook The Center worked directly with the Vermont Agency of Natural Resources to develop a statewide stormwater management design manual as part of Vermont's implementation of a state stormwater program.
Completed: July 2001
Staff Contact: T. Brown

Stormwater Maintenance Manual In this project, the Center is developing a set of web-based tools to help communities develop a state-of the art maintenance program. Stormwater practitioners will be able to download readily usable maintenance forms, updated maintenance cost data, ordinances, and other key elements of a good program.
Staff Contact: T. Schueler

US EPA

EPA commissioned a paper: Opportunities in Nitrogen Management Research: Improving Applications for Proven Technologies and Identifying New Tools for Managing Nitrogen Flux and Input in Ecosystems by Eric E. Jorgensen. The Abstract follows. It identifies 13 tasks in a major research program that, if completed, will provide specific guidance on nutrient management.

"The presence and distribution of undesirable quantities of bioavailable nitrogenous compounds in the environment are issues of long-standing concern. Importantly for us today, deleterious effects associated with high levels of nitrogen in the ecosystem are becoming everyday news events. Excess nitrogen in the environment is associated with many large-scale environmental concerns, including eutrophication of surface waters, toxic algae blooms, hypoxia, acid rain, and global warming. Unfortunately, releases of nitrogen associated with anthropogenic activities are expected to rise throughout the foreseeable future. Whereas our current technologies for managing nitrogen in the environment are stressed, it is reasonable to project that they are likely to fail under the increased loads of nitrogen that are projected for the future. The potential scale of the undesirable consequences is such that it is prudent for us to consider reasonable management and research responses now.

This Issue Paper describes a proposed three-part research and management program that is a measured response to concerns about nitrogen pollution, particularly in the eastern United States. The program describes: 1) steps to be taken with regard to landscape management that will improve our knowledge of nitrogen release and management as it relates to land use; 2) investigations needed that will improve our understanding of the factors that prevent full implementation of nitrogen management technology in the high use landscapes that comprise 35.2% of the land cover in the eastern United States; and, 3) research that is needed to help uncover cause-and-effect relationships among trophic levels that will provide new tools for managing nitrogen, especially on low use landscapes that comprise 64.8% of the land cover in the eastern United States."

Contact Information: Eric Jorgensen (580) 436-8545 at the Subsurface Protection and

Remediation Division of the National Risk Management Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Ada, Oklahoma.

5.0 COMMONLY ASKED QUESTIONS

1) **What is the potential for accumulation of pathogens and their impact on the waterbody?** Pathogen survival is dependent upon distance from source; a 500' buffer zone between the pathogen source and open water should allow for natural degradation of pathogens. Pathogens are not likely to accumulate in or affect the waterbody (unless it attracts waterfowl) but may affect humans. Reed et al. (1988) note that removal of pathogens in aquatic systems is due to natural die off, predation, sedimentation, and adsorption and is dependent upon detention time and temperature. They state "Investigations have shown that the natural aquatic, wetland, and land treatment concepts provide very effective control of pathogens."

2) **What is the potential for algae blooms and eutrophication, drop in DO, and elevated BOD?** Assuming that the N additions are as nitrate at a known loading, and knowing that the rate of denitrification is nitrate limited, then it is possible to manage a system such that most of the nitrate is denitrified rather than taken up in plant biomass. Converting the nitrogen to N_2 reduces the likelihood of eutrophication/low DO. If carbon additions were needed to enhance denitrification in a carbon poor system, the rate of addition would need to be managed to ensure no increase in BOD at the discharge point.

3) **What is the potential for binding of phosphorus, metals, silicon, organics to underlying sediments over time and how might this affect the function and ecological health of the waterbody?** Most contaminants are bound to clays upgradient of the wetland systems. Dissolved organics such as DON are often highly resistant to degradation and are likely to flow out of the system with minimal effect.

4) **Is there a minimum residence time needed for significant nitrogen attenuation in different types of wetlands and waterbodies? How is residence time controlled?** Generally 12 hours is adequate for a managed subsurface flow wetland. Two to three days may be needed for other wetlands and ponds. Nitrate reduction rates are not correlated with vegetation types. Higher rates are associated with high nitrate, available carbon, temperature ~10C or higher, and good distribution of water through the soil matrix. Detention time and nitrate reduction are non-linear --- that is, twice the nitrate loading does not require a doubling of residence time.

5) **What is the potential impact of hydrological changes on vegetation types in freshwater wetlands? What is the potential impact on fishery productivity and other wildlife habitat?** There is potential for change in vegetation in managed system because the soil would likely be saturated all the time. Vegetation near the influent end may become more lush and shade habitat may

develop. Stream modification to reduce the current and pond deepening would affect water temperature gradients, aquatic plant communities, the depth of the muck layer on the bottom, infauna and some fish spawning habitat.

6) **Does nitrogen attenuation take place in the water column or in the transfer of chemical fluxes through the groundwater/pond interface at up gradient and down gradient locations?** Denitrification is limited to anoxic conditions so it rarely occurs in the water column; it occurs at the sediment water interface and in the sediment.

7) **How does flow beneath shallow ponds affect natural attenuation of nitrogen in groundwater?** Flow immediately below pond bottoms, where organic material can diffuse down and where the conditions are anoxic, is the place where natural attenuation occurs.

8) **How does attenuation differ if it occurs as part of a groundwater discharging system, where groundwater flows through sediments versus a groundwater recharging system where groundwater emerges and flows into surface water that may include water column microbial action?** Surface water with microbial action (i.e. microbes performing denitrification in the water column itself) is not common in natural systems but can occur and has been seen in septic treatment systems where septage was discharged directly to surface water (John Teal, personal observation). If the water column has anoxic zones (and all the other conditions necessary for denitrification), microbial denitrification can occur. In general, a groundwater recharge system is more likely to pass through sediments with conditions favorable for denitrification (including anoxic zones, adequate labile organic carbon). Note, however, that the key point is that nitrate-laden water flow through areas with conditions favorable for denitrification. And if groundwater discharge occurs in a suitable pond or wetland, the surface water would eventually be denitrified, whether or not denitrification was occurring in the water column itself. In short, it is unlikely that high rates of denitrification would occur in the water column itself in most places, but if surface water comes in contact with appropriate sediment conditions denitrification can be just as fast as it would be in a groundwater recharge system.

9) **For groundwater discharge systems, how much of the groundwater column must be part of the interaction with the wetland or waterbody for effective attenuation of nitrogen?** Nitrogen attenuation is proportional to the amount of groundwater intercepted and to the nitrate loading.

10) **How does winter dieback of vegetation and subsequent release of nitrogen as it decays affect the long-term attenuation of nitrogen?** The interannual effect is insignificant. The primary removal method is microbial denitrification that occurs throughout the year in groundwater and subsurface soils.

6.0 ACKNOWLEDGEMENTS

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7.0 CITED REFERENCES

Addy, K.L., Gold, A. J., Groffman, P. M., and P. A. Jacinthe. 1999. Ground water nitrate removal in subsoil of forested and mowed riparian buffer zones. *Journal of Environmental Quality* 28:962-970.

Addy, K., Gold, A., Nowicki, B., McKenna, J., Stolt, M., and P. Groffman. 2005. Denitrification capacity in a subterranean estuary below a Rhode Island fringing salt marsh. *Estuaries* 28:896–908.

Ahlgren, I, F Sorensson, T Waara and K Vrede. 1994 Nitrogen budgets in relation to microbial transformations in lakes. *Ambio* 23: 367-377.

Arheimer, B and HB Wittgren. 1994. Modeling the effects of wetlands on regional nitrogen transport. *Ambio* 23: 378-386.

Blazejewski. 2005. *In situ* ground water denitrification in stratified, permeable soils underlying riparian wetlands. *Journal of Environmental Quality* 34:524-533.

Boulton, A.J., S. Findlay, P. Marmonier, E.H. Stanley, and H.M. Valett. 1998. The functional significance of the hyporheic zone in streams and rivers. *Annual Review of Ecological Systematics* 29:59-81.

Crites, R. and G. Tchobanoglous. 1998. Small and Decentralized Wastewater Management Systems. McGraw-Hill, New York

Crumpton, W.G., Isenhart, T.M., and S.W. Fisher. 1993. Fate of non-point source nitrate loads in freshwater wetlands: results from experimental wetland mesocosms. Ch. 29 in Constructed Wetlands for Water Quality Improvement. GA Moshiri, ed. Lewis Publishers.

Davis, J.L., B. Nowicki, and C. Wigand. 2004. Denitrification in fringing salt marshes of Narragansett Bay, Rhode Island, USA. *Wetlands* 24:870-878.

Gleischer, S, A Gustafson, A Joelsson, J Pansar and L Stibe. 1994. Nitrogen removal in created ponds. *Ambio* 23: 349-357.

Groffman, P.M., A.M. Dorsey, and P.M. Mayer. 2005. N processing within geomorphic structures in urban streams. *Journal of the North American Benthological Society* 24:613-625.

Hill, A.R. and K. Sanmugadas. 1985. Denitrification rates in relation to stream sediment characteristics. *Water Research* 19:1579-1586.

Hill, A.R., P.G.F. Vidon, and J. Langat. 2004. Denitrification potential in relation to lithology in five headwater riparian zones. *Journal of Environmental Quality* 33:911-919.

Howard-Williams, C. 1985. Cycling and retention of nitrogen and phosporus in wetlands: a theoretical and applied perspective. *Freshwater Biology* 15:391-431.

Howes, B.L., P.K.Weiskel, D.D.Goehringer and J.M.Teal. 1996. Interception of freshwater and nitrogen transport from uplands to coastal waters: the role of salt marshes. pp. 287-310 in Estuarine Shores: Evolution and Human Alternations, K.F.Nordstrom and C.J.Roman eds. Wiley, London.

Jacks, G, A Joelsson and S Fleischer. 1994. Nitrogen Retention in Forest Wetlands. *Ambio* 23: 358- 362.

Jansson M., R. Andersson, H. Berggren and L. Leonardson. 1994a. Wetlands and lakes as nitrogen traps. *Ambio* 23:320-325

Jansson M., L. Leonardson and J Fejes. 1994b. Dentirification and nitrogen retention in a farmland stream in Southern Sweden. *Ambio* 23:326-331.

Johnston, CA. 1991. Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. *Critical Reviews in Environmental Contamination* 21:491-565.

Kadlec, R., and R. Knight. 1996. Treatment Wetlands, Lewis Publishers. Boca Raton, FL.

Kellogg, D.Q., Gold, A.J., Groffman, P.M., Addy, K., Stolt, M.H., and G. Mayer, P.M., S.K. Reynolds, M.D. McCutchen, and T.J. Can.eld. 2006. Riparian buffer width, vegetative cover, and nitrogen removal effectiveness: A review of current science and regulations. EPA/600/R-05/118. Cincinnati, OH, U.S. Environmental Protection Agency.

Leonardson, L, L Bengtsson, T Davidsson, T Persson and U. Emanuelsson. 1994. Nitrogen retention in artificially flooded meadows. *Ambio* 23: 332-341.

Massachusetts DEP, Environmental Monitoring for Volunteers Guidebook. Available online at http://www.mass.gov/dep/public/volmonit.htm#guidebook.

Moore, J.R.J., P.A. Keddy, C.L. Gaudet, and I.C. Wisheu. 1989. Conservation of wetlands: Do infertile wetlands deserve a higher priority? *Biological Conservation* 47:203-217.

Nowicki, B.L., E. Requintina, D. VanKeuren, and J. Portnoy. 1999. The role of sediment denitrification in reducing groundwater-derived nitrate inputs to the Nauset Marsh estuary, Cape Cod, Massachusetts. *Estuaries* 22:245-259.

Peterson, S.B. and J.M.Teal. 1996. The role of plants in ecologically engineered wastewater treatment systems. *Ecological Engineering* 6:137-148.

Phipps, R.G. and W.G.Crumpton. 1994. Factors affecting nitrogen loss in experimental wetlands with different hydrologic loads. *Ecological Engineering* 3:399-408.

Redfield, A.E. 1972. Development of a New England salt marsh. *Ecological Monographs* 42:201-237

Reed, S.C., E.J. Middlebrooks and R.W. Crites. 1988. Natural Systems for Waste Management & Treatment. McGraw Hill, New York. 308 pp.

Reed, S.C., R.W. Crites, and E.J. Middlebrooks. 1995. Natural systems for waste management and treatment, 2nd Ed., McGraw-Hill, New York

Santrueckova, H; Picek, T; Simek, M; Bauer, V; Kopecky, J; Pechar, L; Lukavska, J; Cizkova, H. 2001. Decomposition processes in soil of a healthy and a declining Phragmites australis stand. *Aquatic Botany* 69(2-4):217-234.

Saunders, D.L. and J. Kalff. 2001. Nitrogen retention in wetlands, lakes, and rivers. *Hydrobiologia* 443:202-212.

Seitzinger, S.P. 1988. Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. *Limnology and Oceanography* 33:702-724.

Silliman, B.R. and M.D. Bertness. 2004. Shoreline development drives invasion of Phragmites australis and the loss of plant diversity on New England salt marshes. Conserv. Biol. 18:1424-1434

Teal, J.M. and B.L.Howes. 1995. Nitrogen balance in a Massachusetts cranberry bog and relationships to coastal eutrophication. *Environmental Science and Technology* 29:960-974

Valiela, I., B. Howes, R. Howarth, A. Giblin, K. Forman, J.M. Teal and J.E. Hobbie. 1982. Regulation of primary production and decomposition in a salt marsh ecosystem. Wetlands: Ecology and Management, Proceedings of the First International Wetlands Conference. New Delhi.

Vought, L.B-M, J. Dahl, C.L. Pedersen and J.O. Lacoursiere. 1994. Nutrient retention in riparian ecotones. *Ambio* 23:342-347.

Weiskel, P.K. and B.L. Howes. 1992. Differential transport of sewage-derived nitrogen and phosphorus through a coastal watershed. *Environmental Science and Technology* 26:352-360.

Weisner, S.E.B., P.G. Eriksson, W. Graneli and L. Leonardson. 1994. Influence of macrophytes on nitrate removal in wetlands. *Ambio* 23:363-366.

White, D.S. and B.L. Howes. 1994. Long-term ^{15}N-nitrogen retention in the vegetated sediments of a New England salt marsh. *Limnology and Oceanography* 39:1878-1892.

APPENDIX A – NITROGEN ATTENUATION BIBLIOGRAPHY