Constructed Wetlands for Treatment of Nursery and Greenhouse Runoff

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Nutrients, copper, and biocides contained in runoff from nursery and greenhouse operations pose a risk to the receiving waters and groundwater drinking supplies. Due to the concentration of these compounds in the wastewater, nursery effluents can be classified as an agricultural waste and the direct disposal of these effluents to the environment is an offence under the federal Fisheries Act and the provincial Waste Management Act. Under correct circumstances, wetlands specifically designed and constructed for water treatment can offer a cost effective, low maintenance water treatment alternative to the nursery and greenhouse industries.

INTRODUCTION

The nursery industry in British Columbia is a growing agricultural sector serving a wide range of consumer requirements from seedlings for silvaculture prescriptions to bedding plants for apartment balconies. To optimize plant production fertilizers, biocides, and growth inhibitors (i.e., $Cu(OH)_2$) are utilized in various capacities. Nursery and greenhouse effluents are unique with respect to other well-characterized and studied wastewaters due to their high nutrient and low organic carbon concentrations. Of particular environmental concern is the nitrate, phosphate, and copper present in runoff from nursery and greenhouse operations and the potential for these compounds to impact the quality and safety of drinking water supplies, induce eutrophication of the receiving waters, and alter the structure and dynamics of aquatic ecosystems. Research conducted at nurseries utilizing controlled-release fertilizers and having leaching fractions near 30% have reported nitrate-N concentrations of 220 to 580 mg liter^{-1} present in the water (Huett, 1997; Yeager and Cashion, 1993). Huett (1997) reported 43% of the nitrogen and 18% of the phosphorus applied as controlled-release fertilizers to potted crops was lost to leaching under normal operating conditions in Australia. Locally, the average nitrate-N and orthophosphate concentrations in the overdrain from vegetable production greenhouses in B.C. is 223 mg liter^{-1} and 99 mg liter^{-1}, respectively (Prystay, 1997). Arnold et al.(1997) reported concentrations of copper between 0.07 and 0.11 mg liter^{-1} in leachate from a nursery utilizing copper-hydroxide-treated containers and suggested that concentrations of 0.5 mg liter^{-1} are possible.

In British Columbia, the regulatory agencies utilize two pieces of legislation, the Federal Fisheries Act and the Code of Agricultural Practice for Waste Management under the Provincial Waste Management Act, to control the discharge of agricultural wastewaters. Under section 36(3) of the Fisheries Act, it is an offence to deposit any deleterious substance into water frequented by fish, including water that may eventually enter water frequented by fish. Part 5, Section 11 and Section 13 of the

Provincial Code of Agricultural Practice for Waste Management states that agricultural waste must not be directly or indirectly discharged into a watercourse or groundwater. Due to the high nutrient content, the runoff from nurseries and greenhouses can be considered an agricultural waste and therefore should be treated prior to discharge.

The use of constructed wetlands for water treatment poses a unique opportunity to the nursery and greenhouse industries as periods of high wetland productivity roughly parallel the production season with the winter dormancy period for wetlands coinciding with winter shut-down and periods of low wastewater production. The beneficial aspects of using wetlands for wastewater treatment include: relatively low construction costs, low maintenance requirements, tolerance to variable hydrological and contaminant loading rates, and reliable wastewater treatment.

WETLAND DESIGN AND REMOVAL PROCESSES

A significant body of treatment wetland research has been published since 1979 assessing the use of wetlands to treat a wide range of wastewaters including municipal sewage, landfill leachate, acid mine drainage, urban stormwater and agricultural wastewaters. These studies have demonstrated constructed wetlands to be effective tools for nutrient management and suggested that they may be more cost-effective than other treatment alternatives (Haberl and Perfler, 1990; Johnston, 1991; Reed and Brown, 1995). During this period three major design strategies have developed: surface flow, subsurface flow, and vertical flow. Of these, surface flow wetlands are the recommended design for greenhouse and nursery operations based on the characteristics of the runoff and cost considerations (Prystay, 1997).

Pollutants entering a wetland system are eliminated through a combination of physical, chemical, and biological processes. With the exception of plant assimilation and the various adsorptive processes, the contaminant processing mechanisms in wetlands are very similar to the removal processes that occur in conventional wastewater treatment facilities. As these processes are naturally occurring in the wetland environment, treatment objectives similar to those established for traditional biological wastewater treatment facilities can theoretically be met at a relatively low developmental and operational cost. The chief drawback to the use of wetlands is related to the natural treatment processes which are inherently slower; longer residence times are required to meet treatment objectives and therefore constructed wetland facilities have large land requirements.

Wetland design is site specific and is dependant upon a number of factors including: volume of wastewater to be discharged; target pollutants to be removed; discharge guidelines; land area available; and, whether wildlife habitat is to be incorporated into the design. The removal of nitrogen, phosphorus, and copper from wastewater occur by very different mechanisms and these all must be promoted to achieve the treatment objective. Nitrate and ammonium nitrogen can be microbially transformed into nitrogen gas and be permanently removed from the nursery effluents under the correct environmental conditions by way of nitrification and denitrification. Phosphorus and copper, on the other hand, are conservative and must be removed by promoting plant uptake, adsorption to sediments and chemical precipitation.

NITROGEN REMOVAL MECHANISMS

Constructed wetlands have been utilized for the removal of nitrogen from wastewater streams with highly variable degrees of success. The predominant removal mechanism is the combined processes of nitrification and denitrification. The conversion of ammonia to nitrite, and nitrite to nitrate, by bacteria is collectively known as nitrification. The group of bacteria which carry out these reactions are generally called nitrifying bacteria or nitrifiers. Nitrification is a two step process, the oxidation of ammonia to nitrite by *Nitrosomonas europaea* and the oxidation of nitrite to nitrate by *Nitrobacter winogradski* (USEPA, 1993).

$$NH_4^+ + {}^3/_2\,O_2 + 6\,e^- \xrightarrow{(N.europae)} NO_2^- + H_2O + 2\,H^+$$

$$NO_2^- + 0.5\,O_2 \xrightarrow{(N.winogradki)} NO_3^-$$

$$NH_4^+ + 2\,O_2 \rightarrow NO_3^- + H_2O + 2\,H^+$$

Denitrification is loosely defined as the reduction of nitrate to a gaseous product, resulting in a loss of fixed nitrogen from the affected environment. The production of nitrogen gas by denitrification is depicted below and is mediated by a number of different microorganisms (Soderlund and Rosswall, 1982; Boyd, 1984).

$$6\,(CH_2O) + 4\,NO_3^- \rightarrow 6\,CO_2 + 2\,N_2 + 6\,H_2O$$

In wetland soils denitrification occurs in anaerobic regions of the litter and below the aerobic sediments. The rate of denitrification is dependant on the supply of NO_3^-, temperature, pH, redox potential, and available biodegradable organic carbon. While in natural systems nitrate is typically found in very low concentrations, many wastewaters have high nitrate concentrations. In these instances temperature and available organic carbon are the limiting factors for denitrification.

The final dominant mechanism for the removal of nitrogen from the wetland environment is the incorporation of nitrogen into plant tissues and eventually the sediments. Biomass analysis of natural and constructed wetlands have shown wetlands to produce approximately 1 ton of above-ground plant biomass annually (Vymazal, 1995). Of this, up to 30% does not decompose over one year and slowly contributes to the development of the sediments (Godshalk and Barko, 1985). Over long periods this accretion will result in the permanent loss of nitrogen from the biological cycle; however, on an annual basis, the amount removed by this route in extremely small when compared to the nitrification/denitrification cycle.

PHOSPHORUS REMOVAL MECHANISMS

Unlike nitrogen, phosphorus is a conservative nutrient and cannot be permanently removed from solution by conversion to a gaseous form. To remove phosphate from the aquatic environment, it must be adsorbed to sediment minerals, assimilated into plant tissues, or precipitated from solution. Retention of phosphate in natural wetland systems has been demonstrated to be directly related to the content of Fe and Al in the sediments (Richardson, 1985). Under acidic conditions PO_4 can be precipitated as insoluble Fe and Al-phosphates, while Ca-phosphate precipitate formation is the dominant removal transformation under neutral to alkaline conditions (Faulkner and

Richardson, 1989). Adsorption to oxides and hydroxides of Al and Fe are also potential removal mechanisms under acidic conditions (Huang, 1980).

The precipitation of phosphorus as calcium phosphate is the most promising method of phosphate removal. Laboratory studies with greenhouse effluents (95 mg liter^{-1} PO_4) demonstrated greater than 90% reductions in soluble phosphate from solution by increasing the pH from 6.5 to 8.0 (Prystay, unpublished data). One method of promoting phosphate precipitation in the presence of calcium, without the addition of a strong base to increase the pH, is to create an environment which will promote a dense algal population and subsequently a high rate of CO_2 uptake. Natural waters are buffered by dissolved inorganic carbon. The status of the CO_2-HCO_3^--CO_3^{2-} equilibrium, with respect to which species is dominant, is pH dependant. The equilibrium state of the buffering system depends on the concentration of hydrogen ion, amount of excess base, the partial pressure of carbon dioxide in the atmosphere, and temperature. When algal biomass growth in a pond is rapid, the uptake of CO_2 from solution by the biomass outstrips the rate it can be replaced and HCO_3^- and CO_3^{2-} are converted to CO_2 producing OH^- which raises the pH of the water (Zehnder 1982, Kadlec and Knight 1996). To promote optimal calcium phosphate precipitation, a pH between 7.5 and 9 will ensure all phosphate is present as dihydrogen phosphate and able to combine with the calcium in solution but not so basic as to promote calcium carbonate precipitation (above pH 9) which would reduce the availability of Ca^{2+} for further phosphate removal.

COPPER REDUCTION

Copper in water is extremely toxic to aquatic biota, even at concentrations as low as 2 μg liter^{-1}. Like phosphorus, copper is a conservative element and must be removed from solution by incorporation into the sediments. This can be effectively accomplished in wetlands due to a number of factors including: the ability of emergent macrophytes to assimilate high levels of copper; a high affinity for copper to bind to organic matter; and, the decreasing solubility of copper with increasing pH. In treatment wetlands planted with *Typha latifolia*, up to 200 mg copper has been assimilated per kilogram plant biomass (Reed et al., 1996). With increasing pH and organic matter the percentage of free ionic copper in solution can be reduced by up to 99%. Additionally, organically bound copper has been established to be nontoxic (CCME, 1995) and constructed wetlands designed with peat substrates have demonstrated up to 98% copper removal efficiencies (Sobolewski et al., 1995).

CONCLUSION

In light of the federal and provincial environmental protection legislation, the nursery and greenhouse owners must address the issue of water quality in the runoff from operations. Constructed wetland systems may offer an ideal wastewater treatment option. Research over the past 16 years has demonstrated constructed wetlands to be effective tools for nutrient management and indicates that they may be more cost-effective than other treatment alternatives. Further, periods of high wetland productivity roughly parallel the production season with the winter dormancy period coinciding with winter shut-down and periods of low wastewater production. By understanding the biogeochemical cycles of target wastewater constituents and the processes that occur within natural wetland systems, treatment wetlands can be designed to take advantage of specific natural processes which improve water quality.

LITERATURE CITED

Arnold, M.A., D.C. Wilkerson, B.J. Lesikar, and **D.F. Welsh**. 1997. Impacts of copper leaching from copper-hydroxide treated containers on water recycling, nursery runoff and growth of baldcypress and corn. J. Amer. Soc. Hort Sci. 122(4):574-581.

British Columbia Ministry of Environment Lands and Parks. 1994. Water quality criteria: Approved and working criteria for water quality. Victoria: MoELP.

Canadian Council of Ministers of the Environment. (CCME) 1995. Canadian Water Quality Guidelines. Ottawa.

Godshalk, G.L. and **J.W. Barko**. 1985. Vegetative sucession and decomposition in reservoirs. pp. 59-77. In: Microbial processes in reservoirs. Dr. W. Junk Publishers, Boston.

Haberl, R. and **R. Perfler**. 1990. Nutrient removal in a reed bed system. Water Sci. Tech. 23(4):729-737.

Huett, D.O. 1997. Fertilizer use efficiency by containerized nursery plants. 2. Nutrient leaching. Aust. J. Agric. Res. 48:259-265.

Huang, P.M. 1980. Adsorption processes in soil. pp. 47-61. In: The handbook of environmental chemistry. Springer-Verlag, Berlin.

Johnston, C.A. 1991. Sediment and nutrient retention by freshwater wetlands: Effects on surface water quality. Crit. Rev. Environ. Control. 21(5,6):491-565.

Kadlec, R.H. and **R.L. Knight.** 1996. Treatment wetlands. Lewis Publishers, Boca Raton, Florida.

Prystay, W. 1997. An assessment of constructed wetlands for the treatment of greenhouse wastewaters, M.Sc. Thesis. University of British Columbia.

Reed, S.C. and **D.S. Brown.** 1995. Subsurface flow wetlands — A performance evaluation. Water Environ. Res. 67(2):244-248.

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

Richardson, C.J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228:1424-1427.

Sobolewski, A. 1996. Metal species indicate the potential of constructed wetlands for long term treatment of metal mine drainage. Ecol. Eng. 6(4):259-271.

USEPA. 1993. Manual: Nitrogen control. USEPA, Cincinnati, Ohio.

Vymazal, J. 1995. Algae and element cycling in wetlands. Lewis Pub., Boca Raton, Florida.

Yeager, T.H. 1993. Controlled-release fertilizers affect nitrate runoff from container plants. HortTechnology 3:174-7.

Zehnder, A.J.B. 1982. The carbon cycle. pp. 83-110. In: The handbook of environmental chemistry. Vol. 1, Part B. Springer-Verlag, Berlin.