Ecological Engineering 36 (2010) 503–516

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Nitrate dynamics in event-driven wetlands

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ARTICLE INFO

Article history:
Received 2 May 2009
Received in revised form 9 November 2009
Accepted 23 November 2009

Keywords:
Treatment wetlands
Nitrate
Events
Dynamic models
Hydraulics
Tracers

ABSTRACT

The dynamics nitrate retention and export were studied at the Des Plaines River wetland demonstration site. Seven wetlands received pulses of river water in discrete pumping events. Twenty-eight wetland events were monitored over 4 years for all hydrologic variables, pumping, rain, storage change, and outflow. Nitrate was measured at high frequency for the ouflows, and at lower frequency for inflows and interior stations. Most events were isolated in time, with sufficient inter-event spacing to allow complete equilibration before the subsequent event. Pumping was selected to provide up to 45 displacements of the wetland water volume. River water averaged 2.3 mg/L of nitrate nitrogen, and wetland effluent averaged 0.9 mg/L. The average mass removal of nitrate was 67% over all events, with a range from 17% to 100%. A calibrated dynamic water mass balance was developed as the framework for interpreting results. Internal hydraulics were characterized by tanks-in-series (TIS) models calibrated to tracer studies. Residence time distributions were describable by three TIS (three wetlands) and five TIS (four wetlands). Dynamic nitrate mass balances were used, in conjunction with a first order areal uptake model, to model the time sequence of NO_3N concentrations and flows. Parameter estimation, based on NO_3N mass flow fitting, produced rate constants that best described the series of events the wetlands. Rate constants were much higher for the events than for previous steady state performance for the wetlands (k_{20} = 107 vs. 37 m/yr). Rate coefficients increased at higher water temperatures, with a modified Arrhenius temperature factor of 1.090. Performance for N removal was found to be partially due to displacement of antecedent treated water, and partially due to treatment occurring during the event, and partially due to treatment after the event. Carbon availability was estimated not to limit denitrification, except possibly at the highest nitrate loadings.

1. Introduction

The nitrogen content of the streams and rivers of the Midwestern United States is of particular importance at this point in history, because of hypoxia in the Gulf of Mexico, together with the associated ecological and economic consequences (Diaz and Solow, 1999). Both point and non-point sources contribute to the nitrogen content of waters within the Mississippi River drainage basin, with about 60% of the water-borne total nitrogen in the form of nitrate (Goolsby and Battaglin, 2000). The source of the nitrogen is about two-thirds from agriculture, and one-third from other sources, including urban runoff, atmospheric deposition, and point sources. A substantial part of the fertilizer nitrogen applied to cultivated crops may be lost in agricultural drainage water. Despite best efforts, it is unlikely that on-farm chemical management alone can solve these contamination problems. The best solutions will involve a combination of on-field and off-field approaches. One of the most promising strategies for reducing non-point source contamination of surface and groundwaters is the use of constructed or restored wetlands specifically as sinks for agricultural chemical contaminants. The premier natural system that has the capability to effectively remove nitrate from surface water is the free water surface wetland, based simply on its ability to place contaminated water in intimate contact with the biogeochemical cycle that removes N (Mitsch et al., 2005).

A large technical database on wetland performance has been accumulated in the broader context of the use of treatment wetlands (Kadlec and Wallace, 2008). In combination with numerous recent detailed studies of wetland biogeochemistry and ecology, a firm basis for the design of continuous flow systems is available. However, there are many situations in which the flow that would enter a treatment wetland would be episodic, because field runoff is associated with rain events. However, it is clear that wetlands differ greatly in their ability to improve water quality, even when compared at equivalent depths, areas, and nominal detention times. For steady flow wetlands, these parameters have included nominal detention time (wetland water volume divided by input volumetric flow rate), hydraulic loading rate (input volumetric flow rate

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doi:10.1016/j.ecoleng.2009.11.020

divided by wetland water surface) and nutrient loading rate (mass of nutrient addition divided by wetland water surface area). For pulse flow, stormwater wetlands, additional parameters include wetland watershed area ratio (Schueler, 1992; Strecker et al., 1992), hydrologic effectiveness (Wong and Geiger, 1997), and stochastic effects (Wong and Somes, 1995).

In contrast to the constant flow wetlands, treatment wetlands in agricultural settings can be expected to receive periodic or pulse inputs of water. These periodic runoff events from fields in agricultural production convey suspended solids, excess fertilizers, and agricultural chemicals from the fields to receiving water bodies. Attempts to correlate wetland performance with very simple design variables such as hydraulic loading, detention time, and pollutant areal loading have generally failed to produce acceptable results. Greater knowledge of internal water movement could help remove a significant amount of this site specificity. It is therefore necessary to examine the internal processes of wetlands.

The overall objective of the work described here is to develop simple models of nitrate loss for wetlands receiving pulsed loads of non-point source nitrate. First, dynamic hydrologic budgets of the several wetlands were determined from precipitation, estimated evapotranspiration, pump-controlled inflow and measured outflow data. Second, conservative tracers were used to determine the residence time distributions of the wetlands. Next, nitrate inflow and outflow concentrations were combined with hydraulic data to determine the mass balances. Finally, the nitrate and hydraulic budgets were combined to create a dynamic model of nutrient removal within the wetlands. Continuous dynamic modeling has been suggested as a means of reducing variability (Konyha et al., 1995), and that option is explored in this paper. This type of model is suited for time series design techniques, with stochastic runoff events as the driving force (Werner and Kadlec, 1999).

2. Site description

Research was conducted at the Des Plaines River Wetlands Demonstration Project site near Wadsworth, Illinois, USA, 80 km north of Chicago (42°26′N, 87°56′W). Upstream of the site, the Des Plaines River drains approximately 520 km^2 of mainly agricultural land (80%) in southeastern Wisconsin and northeastern Illinois. Wetland construction at the site began in 1986, and four wetlands were completed in October 1988. Construction of sedge meadow wetlands were begun in 1991, and completed in 1992. A pump station installed on the Des Plaines River was used to deliver controlled amounts of river water to each of the wetlands, and water levels and discharge were controlled by weirs at the outlet of each wetland.

Wetlands EW3, EW4, EW5, and EW6 differed in size and shape, but were similar in plant community structure (Fennessy et al., 1994) (Fig. 1). Wetland EW3 is a kidney-shaped basin with its deepest zones in the center (Fig. 2). Vegetation was established by allowing volunteer species to grow, with the exception of two introduced species: *Nymphea odorata* (survived) and *Nelumbo lutea* (did not survive). Vegetation in the deeper zones (40–100 cm) was dominated by submerged aquatic species (e.g., *Ceratophyllum demersum*) and floating leaved plants (e.g., *N. odorata*), and occupied approximately 50% of the wetland areas. Vegetation in the shallower zones (0–40 cm) was dominated by emergent macrophytes, principally *Typha* spp., *Scirpus fluviatilis*, and in startup years, *Phalaris arundinacea* (Fennessy et al., 1994). The stage–storage and stage–area relations for EW3-6 are those characteristic of bowl shaped basins (Hey et al., 1994a).

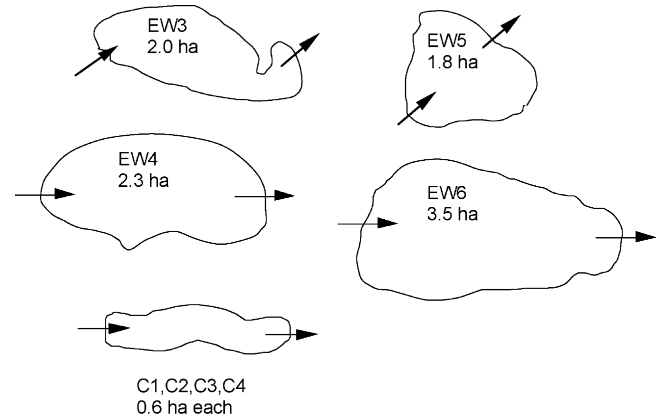

Fig. 1. Sizes and shapes of the test wetlands. EW3, 4, and 5 were approximately 60 cm deep; EW6 and C1–C4 were approximately 60 cm deep. More details on bathymetry are given in Figs. 2 and 3, and in Kadlec (2001). Not to scale.

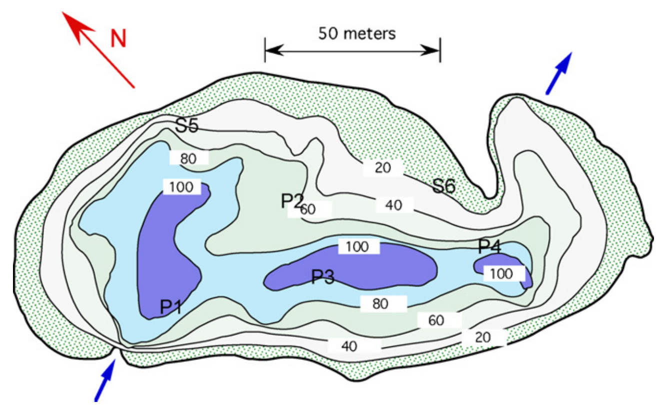

Fig. 2. Bathymetry of wetland EW3. Depths in centimeters. Interior sampling points are also shown.

Wetlands C1, C2, C3, and C4 are morphologically similar (nearly identical) sinuous basins with deep zones at both ends. Wetland C1 is used as an example to demonstrate the basin characteristics (Fig. 3). A channel follows the centerline of each, and benches along the sides provide shallow water when pumping is underway. The intent of this design was to provide seasonal dryout of the benches to foster wetland species that require shorter hydroperiods, such as *Carex* spp. However, vegetation in the deeper zones (ca. 50 cm) became dominated by submerged aquatics, floating leaved plants, and metaphyton. By design, the deep areas occupied approximately 15% of the wetland area. Vegetation in the shallower zones was dominated by emergent macrophytes, principally *Typha* spp. The stage–storage and stage–area relations for C1 are those characteristics of channel and bench basins (Kadlec, 2001).

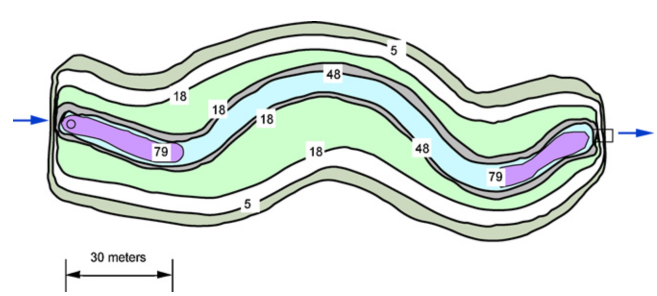

Fig. 3. Bathymetry of wetland C1. Depths in centimeters.