

available at [www.sciencedirect.com](http://www.sciencedirect.com)journal homepage: [www.elsevier.com/locate/ecoleng](http://www.elsevier.com/locate/ecoleng)

# Wastewater treatment at the Houghton Lake wetland: Hydrology and water quality

**Robert H. Kadlec\***

Wetland Management Services, 6995 Westbourne Drive, Chelsea, MI 48118-9527, USA

---

**ARTICLE INFO**

**Article history:**

Received 22 July 2008

Received in revised form

10 October 2008

Accepted 13 October 2008

---

**Keywords:**

Treatment wetlands

Hydrology

Water quality

Nitrogen

Phosphorus

---

**ABSTRACT**

Lagoon-treated wastewater was discharged to a natural peatland to remove nutrients. For thirty consecutive years, an average of 600,000 m<sup>3</sup> of treated water was discharged to the Porter Ranch peatland near the community of Houghton Lake, Michigan. This discharge was seasonal, commencing no sooner than May 1 and ending no later than October 31. During the winter half-year, treated wastewater was stored at the lagoon site. This water contained 3.5 mg/L of total phosphorus, and 7 mg/L of dissolved inorganic nitrogen. Other wastewater quality parameters were CBOD<sub>5</sub> = 15 mg/L, TSS = 34 mg/L, and fecal coliforms at 66 cfu/100 ml. The peatland was large, about 700 ha, but the zone that provided wastewater polishing was approximately 100 ha. Outflows from the larger peatland showed no effects of the discharge, and maintained concentrations of 40 µg/L of phosphorus, and 85 µg/L of ammonia nitrogen. Nutrients were stored in the 100-ha irrigation area, which removed 94% of the phosphorus (53 metric tons) and 95% of the dissolved inorganic nitrogen. All other constituents were also removed in the irrigation area, except for pass-through substances such as chloride. Phosphorus was stored in new biomass, increased soil sorption, and accretion of new soils and sediments, the last being dominant. A simple growth and uptake model described the removal of phosphorus, with an uptake rate coefficient that did not change over time. Thus, rates in this system were stable over time, and the P-removal capacity did not diminish. The irrigation area underwent large changes in ecosystem structure. There was an initial fertilizer response, characterized by much larger standing crops of vegetation. There was also a plant community shift, from the initial sedge-willow cover type to a cattail-dominant cover type. This new cattail patch became a floating mat.

© 2008 Elsevier B.V. All rights reserved.

---

## 1. Introduction

This paper describes the salient water quality and quantity results from a very long-running study of the capacity of a natural peatland to remove nutrients from treated wastewater. Data are presented and analyzed from three decades of full-scale operation, which followed background wetland studies, field mesocosm work, and a 3-year pilot study.

### 1.1. Project description

The community of Houghton Lake, located in the central lower peninsula of Michigan, has a seasonally variable population, averaging approximately 5000. A 2000 m<sup>3</sup>/d sewage treatment plant was built in the early 1970s to help protect the 8100-ha, shallow recreational lake. The wastewater from this residential community is collected and transported to two 2-

---

\* Tel.: +1 734 475 7256.

E-mail address: [rkhkadlec@chartermi.net](mailto:rhkadlec@chartermi.net).

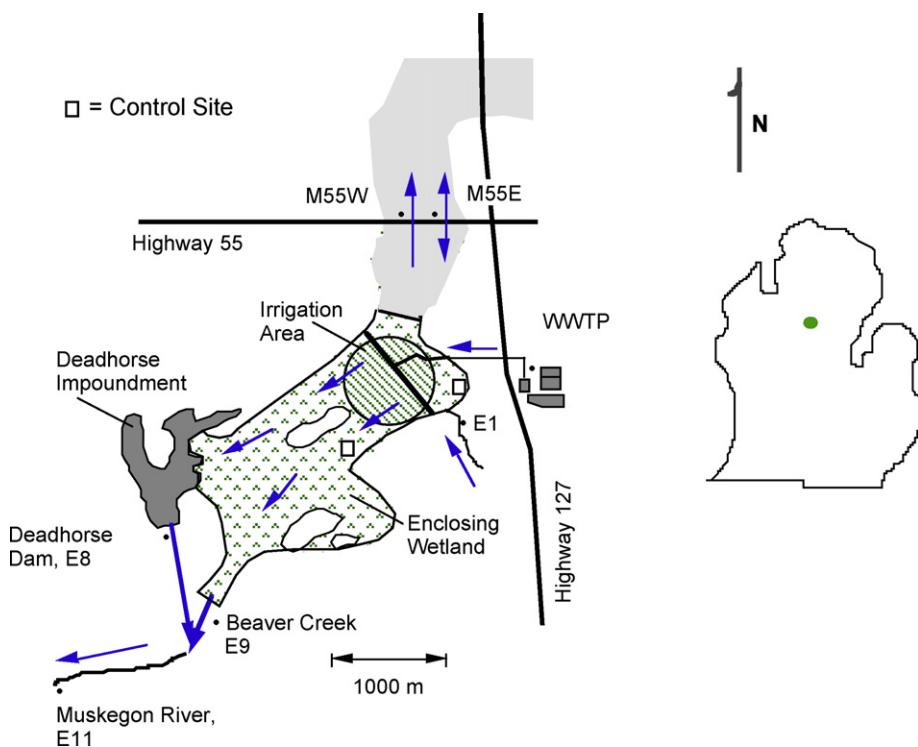
ha aerated lagoons, which provide 6 weeks detention. Effluent is then stored in a 12-ha pond for summer treatment in the Porter Ranch peatland. During each summer season, beginning in May and ending no later than October, stored water is discharged through a dechlorination pond to the peatland. Provision for chlorination is available, but unused because of the low bacterial counts in the storage lagoon. A 30-cm diameter underground force main surfaces at the edge of the peatland, and runs along a wooden boardwalk for a distance of 800 m to the irrigation area in the wetland (Fig. 1). The wastewater may be split between two halves of the discharge pipe, which runs 500 m in each direction. The water is distributed across the width of the peatland through 100 small, gated openings in the discharge pipe, which spread the water over the peatland. Discharges to the wetland ranged from 0.38 to 1.02 million m<sup>3</sup> per year over the history of the project. Pumping is discontinuous, due to maintenance, weekend and holiday shut-offs, or lack of water in the dechlorination pond. Water moves through the wetland to beaver-impounded streams, and then to the Muskegon River.

The Porter Ranch wetland comprises approximately 700 ha. The peatland irrigation site originally contained areas of two distinct vegetation types: one with predominantly sedges (*Carex* spp.) and willows (*Salix* spp.), and a second with leatherleaf (*Chamaedaphne calyculata*) and bog birch (*Betula pumila*). Isolated small patches of cattail (*Typha* spp.) were also present. The edge of the peatland contained alder (*Alnus* spp.) and willow. Standing water was usually present in spring and autumn, but the wetland had no surface water during dry summers. The leatherleaf-bog birch cover type generally had less standing water than the sedge-willow cover type. Soil in the sedge-willow community was 1–2 m of highly decomposed sedge peat, while in the leatherleaf-bog there is 2–5 m of

medium-decomposition sphagnum peat. The entire wetland rests on a clay “pan” several feet thick. Interior flow in the wetland occurs by overland flow, proceeding from the northeast down a 0.02% gradient to a stream outlet (Deadhorse Dam) and beaver dam seepage outflow (Beaver Creek), both located about 3.5 km from the discharge. Wastewater adds to the natural surface sheet flow.

## 1.2. History

The Porter Ranch peatland has been under continuous study from 1970 to the present. Studies of the background status of the wetland were conducted during the period 1970–74, under the sponsorship of the Rockefeller Foundation and the National Science Foundation (NSF). The natural peatland, and 6 m × 6 m mesocosm plots irrigated with simulated effluent, were studied by an interdisciplinary team from the University of Michigan. This work gave strong indications that water quality improvements would result from wetland processes (Richardson et al., 1976, 1978). Subsequently, pilot scale (360 m<sup>3</sup>/d) wastewater irrigation was conducted from 1975 to 1977. This system was designed, built and operated by the Wetland Ecosystem Research Group at the University of Michigan. NSF sponsored this effort, including construction and research costs. The pilot study provided the basis for agency approval of the full-scale wetland discharge system. These preliminary projects produced nine reports to the NSF, fourteen MS and PhD theses, and numerous journal papers. The papers are listed in the Bibliography/References section. Various phosphorus (P) processing mechanisms were studied in the natural wetland prior to the large-scale water and nutrient additions under consideration here (Richardson, 1985; Richardson and Marshall, 1986). The results of these



**Fig. 1 – Location and layout of the Houghton Lake treatment wetland. Arrows indicate direction of flow.**

early background studies pertained to the original, unimpacted ecosystem, and were not descriptive of the transition to the replacement ecosystem or its sustainable functions during full-scale operation. The results of 1975–1977 pilot project studies gave early indications of phosphorus removal potential for moderately large additions of nutrients and water (Tilton and Kadlec, 1979).

The full-scale system was designed jointly by Williams and Works, Inc. and the Wetland Ecosystem Research Group at the University of Michigan. Funding for the project included a 75% U.S. Environmental Protection Agency (USEPA) construction grant. Construction occurred during winter and spring, 1978, with the first water discharge in July 1978. Because the wetland was located on lands owned by the State of Michigan, and supervised by the U.S. Fish and Wildlife Service (USFWS), a condition of use was supplementary research. Compliance monitoring has been supplemented by ecosystem studies, spanning 1978 to present, which have focused on all aspects of water quality improvement and wetland response. Those studies were sponsored by the NSF for 3 years, and for 27 years by the Houghton Lake Sewer Authority (HLSA). Each year, two research reports were generated. One set covered hydrology, water quality, soils and vegetation, and was produced by Robert Kadlec and co-workers. The second set covered animals, ranging from invertebrates to birds and mammals, and was produced by Harold Prince and co-workers. These 60 reports are on file with both the HLSA and the Michigan Department of Natural Resources (MDNR).

The full-scale project has also been the subject of numerous papers, several of which are relevant background for the analysis in this paper. For example, a complex spatially distributed, dynamic, compartmental model of the early phases of the full-scale project was prepared (Hammer and Kadlec, 1986; Kadlec and Hammer, 1988). A summary of the first 15 years of operation was presented in Kadlec (1993). A simplified and revised model of phosphorus removal was presented by Kadlec (1997).

There are two aspects of the project that are of interest: the efficacy of the wetland in providing water quality improvement, and resulting impacts of the wastewater on the ecosystem. This paper addresses the first aspect, water quality

improvement. Three other papers address the second aspect, two dealing with vegetation and soils (Kadlec, 2009; Kadlec and Bevis, 2009) and the other with the animal inhabitants of the wetland (Monfils and Prince, 2009). The water quality performance record is unique, in that it has been continuously acquired over the entire 30-year period of operational record (POR). As a consequence, it is possible to address some of the speculations concerning the long-term capabilities of wetlands to treat water.

### 1.3. Antecedent water quantity and quality

The original peatland was dry during most summers, but had surface water during autumn, winter and spring. In the early twentieth century, attempts were made to drain the system for agriculture, but these met with almost zero success. The drainage ditches exerted little or no influence past a distance of a few meters. Roads were built that cut off surface inflows from the northeast and southeast (Fig. 1), although culverts preserved some of the original drainage. Beaver activity caused cut-off of inflows at interior locations. As a result, the wetland functioned to slowly drain the excess of rain over evapotranspiration, in a southwesterly direction. Water depths in the pre-project peatland ranged from a meter below the peat surface during drought conditions, to 45 cm above the peat surface in the spring melt period of some years. The average summer condition was a dry surface, except for small pockets of standing water. Communication between surface water and groundwater is essentially nil, based on the work of Haag (1979). A clay aquiclude isolates the water under the wetland, which is under considerable hydrostatic pressure. Puncturing this layer results in upward artesian flow.

Extensive water sampling was conducted before any projects were built, both internal to the wetland and at its primary outflows. Measurements were also taken at control locations during the project years. Pre-project nitrate ( $\text{NO}_3\text{N} = 39 \mu\text{g/L}$ ,  $N = 132$ ) and total dissolved P (TDP =  $20 \mu\text{g/L}$ ,  $N = 132$ ) were at low concentrations in surface waters. Control areas were located at places remote from the discharge (Fig. 1). Control area sampling during the project gave total phosphorus TP =  $40 \mu\text{g/L}$  ( $N = 196$ ). Ammonia was found at about

**Table 1 – Pre-project water quality at peatland inflows, outflows and internal locations. Internal sampling was at three depths. Data from unfrozen dates in 1972–1974. There were 19 internal stations on a 400-m grid. Refer to Fig. 1 for inflow and outflow locations. Units are mg/L except pH (s.u.) and EC ( $\mu\text{S}/\text{cm}$ ).**

	Internal wetland			Inflows (M55W)	Outflows (E8/E9)
	Surface	15 cm	45 cm		
NH <sub>4</sub> N	0.728	2.099	1.889	0.200	0.500
NO <sub>3</sub> N	0.039	0.059	0.057	0.035	0.050
TDP	0.020	0.041	0.029	0.040	0.050
Chloride	28	29	23	28	8
Ca	19	30	32	28	8
Mg	3.9	5.6	6.0	5.8	4.9
Na	6.7	5.5	5.1	32	
K	0.7	0.8	0.5	1.1	
Fe	0.20	2.0	1.4	0.19	0.16
pH	6.38	6.23	6.21	7.00	6.50
EC	153	220	225	390	46

0.73 mg/L ( $N=132$ ) pre-project, but was essentially absent in control area samples during the project,  $\text{NH}_4\text{N}=0.085 \text{ mg/L}$  ( $N=284$ ). The disparity may have been an artifact of analytical methodology. Some rainfall dilution may have occurred pre-project, as evidenced by low chloride in the exit stream (Table 1). Seasonal patterns of water quality were minimal during the warm part of the year, but with notable increases in under-ice water. The pH of the wetland surface waters was mildly acidic, with  $\text{pH}=6.3$  ( $N=52$ ) at control locations. The electrical conductivity (EC) was low,  $\text{EC}=111 \mu\text{s/cm}$  ( $N=50$ ) at control locations.

#### 1.4. Operation

Wastewater is collected throughout the year, and treated in the two aerated lagoons. The treated water is then passed to the holding pond, where it is stored during the cold months. The holding pond is gradually emptied during the warm months of the year. Water is transferred first to a dechlorination pond, which acts as the sump for the transfer pump.

The window of opportunity for treatment in the wetland is bounded by the time of the spring thaw and initiation of plant growth, and by the end of plant activity and freeze-up in the autumn. The average patterns of snow, temperature and vegetation growth are shown in Fig. 2. The starting and ending air temperatures were approximately 7–10 °C. The starting and ending dates are formalized in the regulatory permits as May

1 and October 31, giving a 184-day allowable irrigation period. During the first 10 years of operation, the flows from the community, and thus to the wetland, increased from 400,000 to 700,000 m<sup>3</sup> for the year. The amount of water to be sent to the wetland each year remained approximately 700,000 m<sup>3</sup> over the last 20 years. The transfer pump has a capacity of 7000 m<sup>3</sup>/d when its sump, the dechlorination pond, is full. As a consequence, water theoretically needs to be transferred on 100 days out of the 184-day window. However, transfers are often less than the transfer pump capacity, and therefore transfers are often less than the maximum. For the most recent 20 years, the transfer pump was operated an average of 108 d/year. The days of no pumping are determined by the plant operator, and typically included weekends and periods of limited water availability.

## 2. Methods

Wetland monitoring was conducted for purposes of compliance with the NPDES permit, which regulated water quality in discharges to and from the wetland, and a special use permit, concerning the surface water quality within the wetland and in its discharge streams. Water quality was also sampled in support of the ecosystem research required by the special use permit. Water quality sampling and analysis was done independently by the HLSA for compliance purposes, and by the Wetland Ecosystem Research Group (WERG) of the University of Michigan (early years) or by Wetland Management Services (WMS) in later years. Somewhat different protocols were followed by WERG, WMS and HLSA.

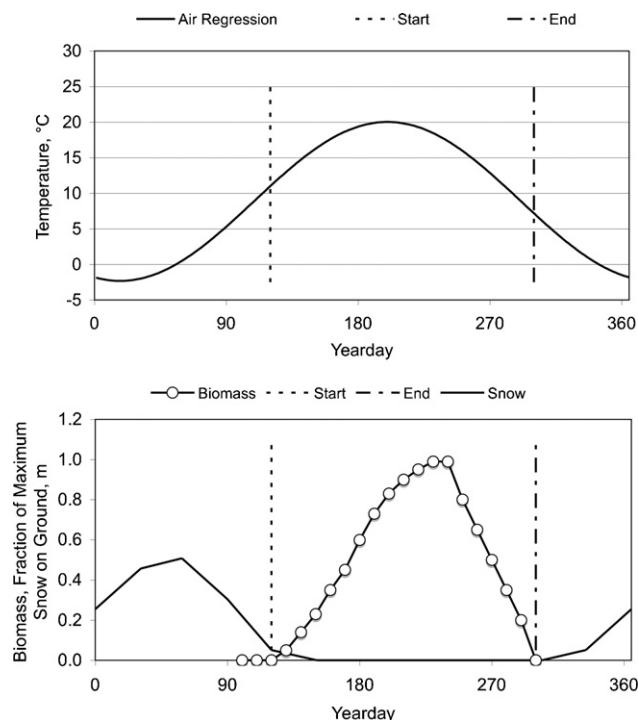
### 2.1. Hydrology and hydraulics

Early in the project lifetime, there were 29 surveyed staff gages located on a rectangular grid covering approximately 100 ha that enclosed the distribution pipeline (see Fig. 3). Additionally, four continuous stage recorders were used, at two upgradient locations and two downgradient locations. Continuous recording ceased in 1986, but staff gages at those locations continued to return data until they fell into disrepair over the next 20 years. Likewise, the original 29 stage gages gradually became non-functional, and in 1988 a new line of six stage gages was surveyed in place along a backgradient transect. Some of these were replaced during 1991 and 2000, and five were replaced in 2006. These locations were also surveyed for ground elevation on eight occasions from 1972 to 2005, and thus water depths could be determined by difference.

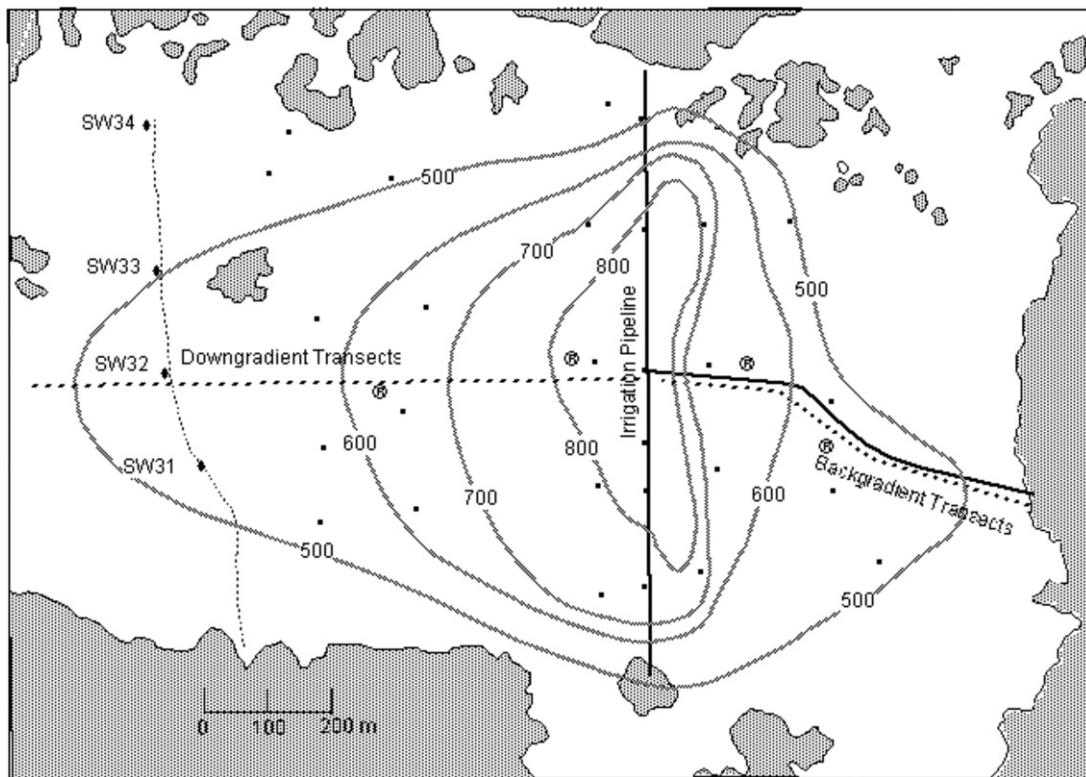
Daily pumping records were maintained by HLSA throughout the duration of the project.

### 2.2. Meteorology

A National Oceanic and Atmospheric Administration (NOAA) weather station is operated on the Porter Ranch, identified as Houghton Lake 6 WSW. It is located approximately 5 km from the discharge pipe, and provided daily records of maximum and minimum air temperatures, as well as daily precipitation. Evapotranspiration was computed from pan evaporation data



**Fig. 2 – Air temperature, snow and vegetation growth patterns at the Porter Ranch treatment wetland. The growth of cattails begins in early May, and senescence is complete by the end of October (lower panel). Regional snow accumulation typically begins in November, and melt is complete in early May. The permitted irrigation season is from May 1 (yearday 120) through October 31 (yearday 304).**



**Fig. 3 – The irrigation area of the Houghton Lake treatment wetland. The estimated contours of electrical conductivity on 9/2/83 ( $\mu\text{S}/\text{cm}$ ) are shown. There were 29 surveyed staff gages (dots), four stage recorder stations (®), and four cross-flow sample stations SW31–SW34.**

at Lake City NOAA weather station, 28 km distant, multiplied by a pan factor of 0.8. This estimate was confirmed by extensive studies of evapotranspiration at the site (Scheffe, 1978; Kadlec et al., 1987a).

### 2.3. Sampling

Water samples taken from the various wetland inflows and outflows were obtained from the flowing streams, and in general presented no problems of entrained extraneous materials. Stream sampling was conducted on a monthly frequency during the irrigation season. Samples were acquired on transects internal to the wetland (Fig. 3), on a monthly frequency during irrigation, and less frequently during the cold months. Water samples from the internal shallow wetland environments were difficult to properly acquire, and few published protocols existed (Nearhoof, 1996). The bottom sediments of the marsh were very easily disturbed, and, hence, easily suspended by foot travel, and by bottle dipping. Sediments and flocs were often of near-neutral buoyancy, and movable by small water currents, such as those created by the flow of water into a sample bottle near the bottom. The water surface may contain floating debris, small floating leafed plants (e.g., duckweed), and filamentous algae. These materials will flow with surface water into a partially immersed sample bottle. The shallower the water, the greater the potential for imbibing bottom sediments and/or surface materials.

For approximately the first 10 years of the project history, surface water was often at depths of 10–20 cm or greater. Water samples could be dipped with little or no entrained material, and therefore no special precautions were taken. However, during years 10–30, there was considerable floc and duckweed in many locations, and dipped samples often contained unrepresentative amounts of those materials.

In recognition of these difficulties, year 10–30 WMS sampling attempted to exclude as many artifacts of the sampling procedure as possible. Pre-cleaned (acid washed) half-liter bottles and caps were rinsed three times with water from an adjacent location. The bottle was immersed and tipped just past horizontal, allowing the bottle to fill very slowly. Nevertheless, extraneous sediments could frequently not be entirely excluded. Consequently, the 500-mL parent samples were allowed to settle for a period of 2–4 h, and the supernatant decanted into 250 mL bottles. The decantation was through a 1-mm (1000  $\mu\text{m}$ ) prewashed stainless steel screen, to remove duckweed and other relatively large objects. Microscopic examination of the undisturbed suspended materials indicated that in general, the upper end of the size distribution was about 200  $\mu\text{m}$ , and thus the screen passed the suspended material. Settling column studies were performed to establish that suspensions of wetland sediments, from both discharge and control locations, completely settled in that time period (Kadlec, 2003).

Marsh samples taken by HLSA personnel were not subject to this protocol. For years 1–10, no special sampling protocols

were followed, and samples were analyzed immediately in a laboratory at the treatment plant. For years 11–30, samples were taken by pipette from clear water pockets near the designated location. The samples were preserved as necessary, and refrigerated for shipment to a USEPA-accredited commercial laboratory. The HLSA was responsible for water quality for compliance purposes, which included weekly analyses of the holding pond discharge water, monthly analyses of four marsh interior stations (SW31–SW34, Fig. 3), and monthly analysis of three wetland stream discharges (E8, E9, E11, Fig. 1).

There is no standard or commonly accepted method for sampling suspended solids in wetland environments. The combination of shallow water and easily disturbed sediments causes unacceptable performance of "standard" samplers and sampling procedures. These problems led to the implementation of a passive sampler, which could be activated after the placement disturbance subsided. This device consisted of a 5-cm diameter opaque plastic tube 60 cm long with a butterfly valve at each end. The valves were constructed to close with minimum force in one-quarter turn of the wire handle. A one-cm drain hole, with a rubber stopper, provided for removal of the sample after the device was removed from the water sheet. Removable, slip-fit end pieces were filled with water and frozen in the laboratory, forming end caps. A wire cross inside the end piece prevented floating of the partially melted ice plug. Side rings on the tube body were used to firmly position the sampler on stakes driven into the peat soil.

In field operation, the stakes were driven into the sediment at the proper location, using the empty tube as the positioning fixture. The ice-filled end caps were added with the valves open, and the tube then filled with solids-free water, through the drain hole. The sampler was then placed on the support stakes and clipped in place, parallel to flow at the desired depth. After placement, the site was left so that the disturbed sediments could resettle and be flushed downstream. Since water velocities exceeded the melting time and settling times were shorter than the melting time, the tube did not open to flushing until the disturbances had subsided and moved downstream.

After flushing for approximately 1 h, the tube was carefully approached from the downstream side, and the valves closed. The entire assembly was then immediately removed and the contents transferred to a sample bottle. The samples were subsequently filtered and dried in the laboratory, following standard methods (APHA, 1975).

#### 2.4. Analysis

Field measurements of pH, EC, temperature and redox potential were made using hand-held meters.

The settled, screened samples were preserved via freezing (years 0–25), and analyzed at the University of Michigan for NH<sub>4</sub>N, TDP and TP, chloride, pH and EC. Nitrate nitrogen (NO<sub>3</sub>N) was analyzed on a small subset of samples. Freezing has been found to adequately preserve nutrients (Marvin and Proctor, 1965; DeGobbis, 1973; Klingaman and Nelson, 1976; Avanzino and Kennedy, 1993; Gardolinski et al., 2001). The freezing method of preservation was checked on a variety of samples, and found to be acceptable for all parameters, except for samples with high levels of total suspended solids

(TSS) or dissolved organic carbon (DOC), which was also confirmed in the literature (MacDonald and McLaughlin, 1982; Fellman et al., 2008). Ammonium, NO<sub>3</sub>N, chloride, EC and pH were measured using specific ion electrodes. Total phosphorus was determined using a colorimetric (ascorbic acid) technique, after digestion of the decanted water sample with sulfuric acid. In years 26–30, analyses were done by commercial, USEPA-accredited labs on acid preserved samples where necessary.

Samples taken by HLSA were analyzed according to Standard Methods by wastewater treatment plant personnel during years 1–10. For years 11–30, analyses were done by commercial, USEPA-accredited labs. Permit parameters for the holding pond varied slightly from year to year, including some or all of: NH<sub>4</sub>N, oxidized nitrogen (NO<sub>x</sub>N = NO<sub>2</sub>N + NO<sub>3</sub>N), TP, TDP, TSS, biochemical oxygen demand (BOD<sub>5</sub>), carbonaceous biochemical oxygen demand (CBOD<sub>5</sub>), chloride, alkalinity, hardness, fecal coliforms (FC), pH and dissolved oxygen (DO). Permit parameters for the internal marsh stations and edge discharge stations were the same, except for FC and TSS.

On a number of occasions, duplicate samples were analyzed via HLSA and WMS procedures. Agreement between labs was generally acceptable, but the unsettled HLSA samples generally produced higher values than WMS settled samples in the low concentration ranges for total phosphorus. For instance, four duplicates averaged 36 µgP/L (MDL = 10) unsettled for HLSA, whilst they averaged 5 µgP/L (MDL = 4) settled for WMS. However, three duplicates averaged 43 µgP/L (MDL = 10) settled for HLSA, whilst they averaged 53 µgP/L (MDL = 4) unsettled for WMS.

A subset of samples was analyzed for sodium (Na), potassium (K), calcium (Ca) and magnesium (Mg), using atomic absorption spectroscopy. Trace metals were checked on two occasions, utilizing neutron activation analysis.

#### 2.5. Averaging

Two kinds of averages are reported here. The average of a set of measurements is accorded the usual meaning, and applies to a specified (POR). It is termed simply the average of the set. The second is the average of annual averages, here termed the annuals average. This measure allocates equal weight to each operational year's mean performance. Because there were uneven numbers of measurements within each year, there is a difference between the POR average and the annuals average, but it is usually slight. The differences in numbers of measurements from year to year were due in major part to differences in the length of the irrigation season, and at some locations to the absence of water to be sampled. Both types are reported as mean ± one standard deviation.

### 3. Hydrology

The wetland is considered to be comprised of two parts: the irrigation area ranging in size up to about 100 ha (Fig. 3), and the entire wetland, encompassing about 700 ha (Fig. 1). There is no physical boundary between these areas. The quantity and quality of the discharges from the entire wetland are of interest for some regulatory purposes. But the irrigation

area is the zone in which many water quality parameters are altered by water passage through the ecosystem, and hence this zone is important to understanding the rates of water quality improvement processes.

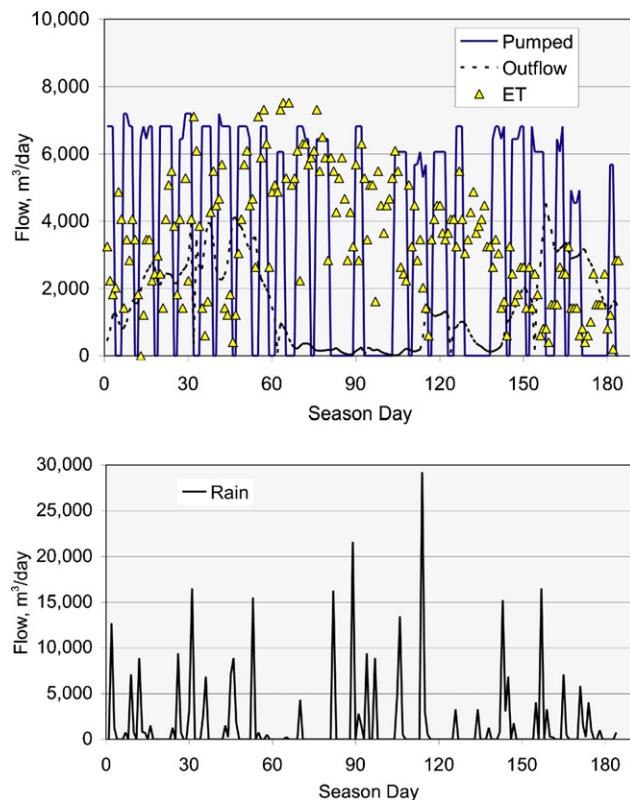
The outflows from the entire wetland are conditioned by a weir (E8) in a man-made levee that creates the Deadhorse impoundment, and by the remnant agricultural drainage ditches (E9), both at the southwesterly end of the wetland (Fig. 1). However, these old features are at the mercy of a large beaver population (Maguire, 1974). Therefore, the weir at E8 has often been completely dammed, stopping flow and diverting wetland drainage to E9. The beavers also dam E9, as well as the entire wetland width of 0.5 km, but the E9 dams are prone to leakage. Outflows at E9 are confined to one to three channels, depending upon beaver activity in dam repair. From time to time, the MDNR removes beaver dams at E8, and removes the animals. Outflows were estimated from channel geometries and time-of-travel measurements of water velocities over the period 1986–2007, and found to average 3750 m<sup>3</sup>/d during the pumping season. Over the same period, the pumped water was 4740 m<sup>3</sup>/d and the net outflow from the irrigation zone was 3530 m<sup>3</sup>/d. Although the outflow estimates are very crude, they do confirm the likelihood that the added water exited in the monitored channels.

Flows into the wetland from the northeast occurred historically (M55E&W, Fig. 1). During the design of the full-scale project, there was a proposal to construct a levee northeast of the irrigation area, spanning the wetland from shore to shore, to block this potential flow. Beavers built essentially this same blockage just prior to project inception. Therefore, the area to the north of the beaver dam and south of Highway M55 is isolated from the irrigation area, and fills and drains in response to water levels north of highway M55. Culvert M55E flowed north (outflow) 50% when flowing, but only flowed 14% of the time. Maximum observed flows were on the order of 50 m<sup>3</sup>/d, or less than 1% of the pumping rate. Culvert M55W flowed north (outflow) 95% when flowing, and flowed 60% of the time. Maximum observed flows were on the order of 100 m<sup>3</sup>/d, or about 1.5% of the pumping rate. Inflows to the wetland also are possible at a road culvert (E1, Fig. 1), but are very infrequent and small. Inflow occurred sporadically at station E1, about one-third of the time, mostly during the early spring, before pumping started. During one-third of the years, zero inflow was observed over the entire monitoring season. The spring flows, when these occurred, were restricted to less than about 100 m<sup>3</sup>/d, or about 1.5% of the pumping rate.

The beaver dams and upland surroundings create a wetland hydrologic unit of about 400 ha (Fig. 1). This area, termed the “enclosing wetland”, includes the irrigation area of about 100 ha, but extends considerably further downgradient than the effects of wastewater addition. In the following discussion, most emphasis will be placed on the irrigation area, or “action zone.” The outflows from the enclosing wetland are of interest because they represent additions to the receiving streams.

### 3.1. Irrigation area water budgets

The irrigation area for the water budget is a 100-ha zone enclosing the distribution pipe. The width of the wetland is approximately 1000 m, and thus the length of the irrigation

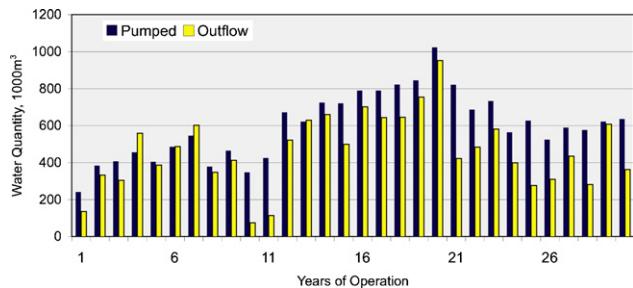


**Fig. 4 – Example water daily budget components for 2002. The irrigation season was from May 1 to October 31. The net water loss for this year was 52%, and resulted in essentially zero outflow in July and August.**

zone is also 1000 m. It extends about 300 m backgradient from the discharge pipe, and 700 m downgradient. As a result of the near-isolation of the irrigation area, it is possible to construct water budgets with a minimum of uncertainty. Flows were all highly variable throughout an irrigation season (Fig. 4). The pattern of rainfall was stochastic, and that of evapotranspiration (ET) was seasonal plus stochastic with a maximum in July (season days 60–90). Pumping was intermittent, with most occurring on weekdays. As a result, outflows, determined by difference, varied from periods of no flow, typically in mid-summer; to periods of moderate flow, typically in spring and again in autumn.

The daily water budgets may be cumulated to an annual budget for each project year, with the resulting inflows and outflows shown in Fig. 5.

Evapotranspiration averaged 3.7 mm/d during the irrigation season (range 2.6–5.0). Evapotranspiration during the months of May through October was calculated to be 82% of the average pumping rate to the 100-ha irrigation area. Rainfall compensated for some of the ET loss, and averaged 59% of the pumped water. Net water losses to the atmosphere (ET-P) were thus 22% of the pumped water. However, some dry years and months caused the wetland to be dry at downgradient compliance sampling locations, about 800 m from the discharge (Fig. 3). For instance, during the period 1997–2007, stations at 800 m were too dry to sample 44% of the time (29/66 monthly cross-transects).



**Fig. 5 – Annual pumping to, and outflow from, the irrigation area. Note that a steady increase in wastewater volumes occurred over the first 21 years. In year 22, repairs to the collection system eliminated a considerable amount of extraneous infiltration.**

### 3.2. Flow model

The stage and discharge at the outlet end of the irrigation area were calculated based upon research on overland flow in the system (Kadlec et al., 1981; Hammer, 1984; Hammer and Kadlec, 1986; Kadlec, 1990). Although the flows to the wetland were episodic, and gave rise to spatially and temporally variable water stages and depths, a reasonable estimate of the outflow from the irrigation area can be made based upon a dynamic level pool analysis. A brief description of the requisite outlet stage-discharge relation follows.

At the outlet of the irrigation area, there was no outlet structure; only a gentle constant ground slope that continued for over a kilometer past the treatment zone. The water depth for a particular flow rate was presumed to be determined by the frictional characteristics of the vegetation, combined with that ground slope. At this site, a friction equation was determined to be:

$$Q = aWh^bS^c = [aWS^c]h^b \quad (1)$$

where  $a, b, c = \text{constants}$ ;  $h = \text{water depth (m)}$ ;  $Q = \text{flow rate (m}^3/\text{d)}$ ;  $S = -dH/dx = \text{negative of the water surface slope (m/m)}$ ;  $W = \text{width of the wetland (m)}$ .

Therefore, the resulting stage-discharge relation is a power law in depth. As an approximation, backed up by field observations, it is assumed that the water surface parallels the ground surface, which had a measured slope of about 0.00002 m/m in the irrigation area. The parameter group  $[aWS^c]$  was selected so that a flow rate of 10,000 m<sup>3</sup>/d was conveyed at a depth of 15 cm. The depth exponent was chosen to be  $b = 3.5$ . Therefore:

$$Q = 10,000 \left[ \frac{h}{0.15} \right]^{3.5} \quad (2)$$

This rule drains water from the irrigation area very easily at high depths, and with great difficulty at low depths. An example of the results of the use of this relationship in daily water mass balancing is shown in Fig. 4, where the outflow rate has been calculated on a daily time step. Importantly, this dynamic mass balance also allows approximate forecasting of water depths.

### 3.3. Irrigation area water stages and depths

Water depths are computed as the difference between the water surface elevation (stage) and the soil elevation. Water surface elevation was determined via recorders and staff gages, and both methods are fairly accurate. However, soil elevation in a peatland is often poorly definable, and the Porter Ranch wetland was no exception. In this project, the definition of the top of the soil was determined by a standard surveyor's staff placed firmly, approximated by a few kilograms weight on a 25-cm<sup>2</sup> bearing area. This definition amounts to selecting the top of the root mat. A 1.25-cm diameter pipe could easily be driven through the entire peat column down to the clay underneath (1.25 cm<sup>2</sup> bearing area; depths up to 2 m of peat). In winter conditions, the top of the soil was taken to be the top of the frozen wet peat.

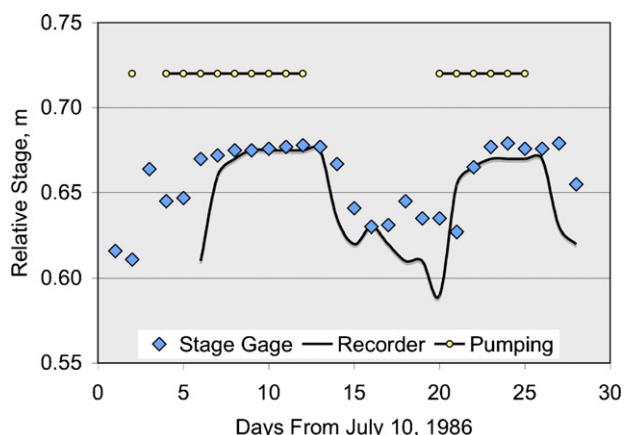
#### 3.3.1. Dynamics and detention

Water depths in the discharge area were generally in the range of 10–20 cm during 1978–1987. The water sheet thinned with increasing distance from the discharge. During the first decade of operation, when the pump was turned on, with water present in the peatland, water stages at the discharge rose only 5–10 cm, and fell a comparable amount when the pump was turned off (Fig. 6). This is reflected in the large depth exponent on the conveyance capacity of the wetland (Eq. (2)). For similar reasons, a large rainstorm did not flood the peatland to great depths.

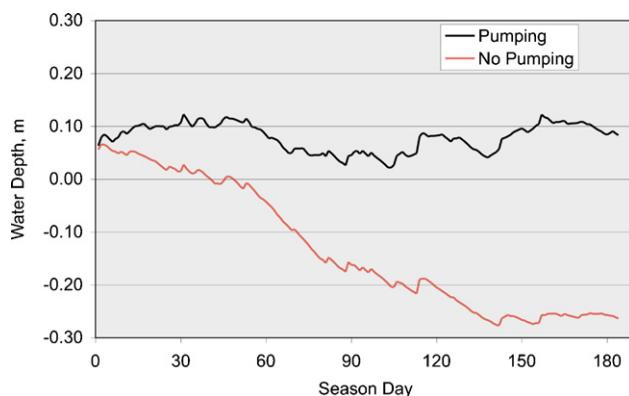
The seasonal average pumping rate was 4670 m<sup>3</sup>/d, equivalent to an average instantaneous pumping rate of 6470 m<sup>3</sup>/d and a 73% duty fraction. Duty fraction is the fraction of the time that the pump is operating. For an estimated mean water depth of 15 cm, the 30-year average nominal detention time was 32 days in the 100-ha irrigation zone, and the long-term average hydraulic loading rate due to pumping was 0.47 cm/d.

#### 3.3.2. Hydroperiod

The hydroperiod of the natural wetland was altered in the zone of discharge; dryout no longer occurred in the irrigation zone, even under drought conditions. The water budget model may be exercised for the rain and ET losses of a particular year,



**Fig. 6 – Response of the wetland water depth to pumping events at the tee in the discharge pipeline. The increase in depth due to pumping, maximum at this location, is ±8 cm.**



**Fig. 7 – Approximate average water depths for 2002, computed from mass balances. The “no pumping” scenario is based upon rain and ET only. The pumping season began May 1.**

but without the addition of water. This simulation of the hypothetical “no pumping” case allows exploration of the effects of the added water (Fig. 7). The pre-project, summer season hydroperiod was about 60–90 days out of a possible 184, or 30–50%. However, at times the water was too shallow to flow or sample at the outlet edges.

### 3.3.3. Long-term effects

Several years after startup, the root mat started to float in the immediate area of the discharge. Initially, the floating mat contained both sedges and cattails, but the sedges disappeared after 8 years, leaving a floating mat of cattail. The location of floating mat was observable by applying surface pressure, which easily would cause visible vertical movement. Sections of the mat showed an approximate 30-cm thickness, containing an interwoven mesh of roots and rhizomes together with considerable quantities of sediments and soil. This mat is discussed in more detail in Kadlec (2009).

By 2002, the floating mat occupied 27 ha of the 100-ha irrigation zone, centered on the distribution pipe tee. Water flow was entirely under this mat, leaving the mat surface as a zone of damp or dry litter. Standing water, in the usual sense of par-

tially immersed plant stems, no longer existed in this central zone.

After approximately 10 years of operation, the power law model (Eq. (1)) was no longer applicable to the immediate vicinity of the discharge. However, the mat did not extend to the edges of the irrigation zone, and consequently the stage discharge relation (Eq. (2)) remained applicable for calculating outflows from the irrigation zone. Within the mat zone, there was no exposed surface water, and hence water loss to the atmosphere was entirely by transpiration.

The fact of under-mat flow meant that the water was exposed to the root zone of the plants, rather than the stem zone. The water quality implications of this mode of flow are not fully understood.

## 4. Input/output water quality for the enclosing wetland

The 400-ha enclosing wetland (Fig. 1), together with its inflows and outflows, may be considered to be a single large system from the viewpoint of the water quality exported to receiving waters. Accordingly, the water quality entering and leaving this large wetland system is first examined. The occasional flows at E1 and M55 had water quality at approximately wetland background values, and did not contribute appreciably to the enclosing wetland because of the extremely low volumes.

### 4.1. Lagoon water chemistry

The combination of aerated lagoons and the six months detention in the holding pond produced a relatively high quality effluent (Table 2). The water quality was approximately secondary, with  $\text{CBOD}_5 = 15 \pm 4 \text{ mg/L}$  (mean  $\pm$  s.d.) and  $\text{TSS} = 33 \pm 22 \text{ mg/L}$ , as the average of 30 annual averages measured by HLSA. There was some nitrogen reduction in the lagoons, with the effluent containing  $7 \pm 4 \text{ mg/L}$  of dissolved inorganic nitrogen ( $\text{DIN} = \text{NH}_4\text{N} + \text{NO}_x\text{N}$ ). Phosphorus was subject to little reduction, and the effluent contained  $3.5 \pm 2.0 \text{ mg/L}$  of TP. Chloride in the lagoon discharge was considerably higher than the wetland background values,

**Table 2 – Thirty-year averages of annual averages of input and output concentrations for the enclosing peatland. Refer to Fig. 1 for locations. Units are mg/L except pH (s.u.), EC ( $\mu\text{s}/\text{cm}$ ) and F. coliforms (cfu/100 ml).**

	HLSA			WMS/WERG		
	Pumped	E8	E9	Pumped	E8	E9
NH <sub>4</sub> N	6.46	0.169	0.316	3.86	0.119	0.078
NO <sub>x</sub> N	1.62	0.160	0.203	1.14	0.140	0.146
Total P	3.51	0.072	0.071	1.79	0.045	0.036
BOD	21.1	3.6	3.6			
CBOD	14.5	2.0	1.8			
Conductivity				693	85	169
Chloride	123	11	25	153	11	34
pH	7.96	6.82	6.54	8.08	5.67	6.74
F. coliforms	66					
TSS	34.3					
DO	6.63					

at  $123 \pm 36$  mg/L in the lagoon effluent compared to about 28 mg/L in the pre-project wetland.

Algae were present in both the holding pond and in the dechlorination pond, and consequently the pH was about  $8.0 \pm 0.3$  and the DO was  $6.6 \pm 0.4$  mg/L (73% saturation). Algae were the cause for elevated TSS. The long detention time, aerobic conditions and long exposure to ultraviolet radiation produced fecal coliforms at  $85 \pm 53$  cfu/100 mL, below the limit of 100 cfu/100 mL that would have required disinfection before discharge.

Samples were also analyzed by WERG/WMS, but at much lesser frequency. These samples typically showed lower levels of nitrogen compounds and phosphorus species, but higher EC and comparable pH (Table 2). Differences in sampling protocols and timing, and analytical methods, may have caused some of the differences.

#### 4.2. E8/E9 chemistry

The principal outflows from the enclosing wetland were at stations E8 and E9 (Fig. 1). The flow proportions varied from time to time, depending on beaver activity. Both stations were monitored for water quality, by HLSA and WERG/WMS. The water quality at these stations was essentially unaltered from the pre-project condition, as seen in Tables 1 and 2. For example, phosphorus left the enclosing wetland at an average of  $0.041 \pm 0.021$  mg/L (mean  $\pm$  s.d., WMS) over the POR, compared to the pre-project average of 0.040 mg/L. There was no time trend in the concentrations leaving the enclosing wetland (Fig. 8). HLSA and WMS/WERG data differ somewhat, but the percent concentration reductions are the same for phosphorus for both groups (98%). However, percentage reduction may be misleading, since the removal was to background levels.

Chloride was a marker for the wastewater, because of the high concentration compared to the antecedent wetland (ca. 140 mg/L vs. 28 mg/L). The concentration of chloride did increase slightly at the E8/E9 outflows, from 8 mg/L pre-project, to 14 mg/L during the first 6 years, and to 20 mg/L during the last 6 years. However, all of these values are less than the interior wetland pre-project average of 28 mg/L (Table 1).

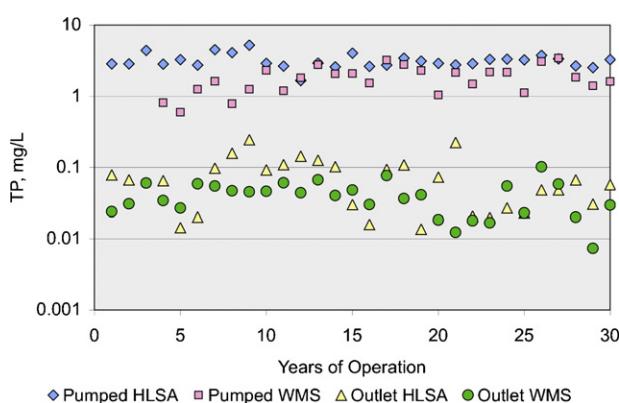


Fig. 8 – Total phosphorus entering and leaving the enclosing wetland. The outlet TP is the average of E8 and E9 for the year.

The lack of a clearly observable response in either interactive or non-interactive substances, phosphorus and chloride respectively, is not surprising because of the large buffering capacity of the Deadhorse impoundment, and the various large beaver impoundments. The turnover time of these water bodies was estimated to be on the order of 3 years, and ice formation provides a mechanism for constituent removal, as discussed below.

Input–output analysis of the enclosing wetland therefore does not provide information on rates of removal, because water leaves at essentially wetland background conditions. Rate information may be determined by focusing on a smaller zone, which contains the gradients of water quality.

#### 5. Irrigation area water quality

The irrigation area has nominal dimensions of  $1000\text{ m} \times 1000\text{ m}$ , with the distribution pipe located about 250 m from the upgradient (northeast) end (Fig. 2). It has been shown that distance along transects is linearly related to the area upstream of the corresponding isopleths of water quality and ecosystem character (Kadlec, 1997). Thus the areas of concentric zones of different water quality are represented here by the corresponding distance from the discharge in the downgradient direction. The relations used were:

$$\text{Downgradient } A = 808x \quad (3)$$

$$\text{Upgradient } A = 1500x \quad (4)$$

where  $A$  = enclosed area ( $\text{m}^2$ );  $x$  = distance along transect from discharge (m).

These allow a one-dimensional analysis (distance) of the performance of the irrigation zone. This 100-ha zone encloses the gradients measured along transects, and its downstream boundary contains four water quality monitoring stations across the flow direction.

Within the 100-ha irrigation area, there was an impacted vegetation zone of lesser size, which increased in size over the course of the project. At year 30, this zone was 80 ha, according to a smoothed model of the impact area increase. Within this 80-ha impact area, phosphorus concentrations were reduced by 98% with respect to background, over the period of record.

Ammonia nitrogen was reduced in a smaller, 50-ha area, with concentrations reduced by 98% with respect to background, over the period of record in a 50-ha area.

##### 5.1. Nutrient loading and removal

###### 5.1.1. Phosphorus

The lagoon discharge added a total of 56.21 metric tons (t) of phosphorus to the wetland irrigation area over the 30-year period. A total of 3.37 t were exported, thus creating a 94% mass removal. This calculation does not include atmospheric sources, which are estimated to be very small (Table 3). The mean concentration reduction for TP was also 94% (Fig. 8). The instantaneous rate of TP addition is the mass of phosphorus added per unit area in a given year, divided by the length of the pumping season in years. On an instantaneous basis, an

**Table 3 – Period of record (POR) flows and loads of DIN and TP to and from the 100-ha irrigation area. Instantaneous rates are based on the length of the pumping season.**

	Water	Phosphorus		Nitrogen	
	Pumping season 1000 m <sup>3</sup> /year	POR gP/m <sup>2</sup>	Instantaneous rate gP/m <sup>2</sup> year	POR gN/m <sup>2</sup>	Instantaneous rate gP/m <sup>2</sup> year
<b>Inputs</b>					
Inflow	598	56.21	5.41	135.1	12.46
Rain	353	0.29	0.03	6.4	0.65
Total	951	56.5	5.44	141.5	13.11
<b>Outputs</b>					
Outflow	488	3.37	0.30	3.4	0.31
Evapotranspiration	463				
Total	951	3.37	0.30	3.4	0.31
Net removal		52.8	5.14	131.7	12.80

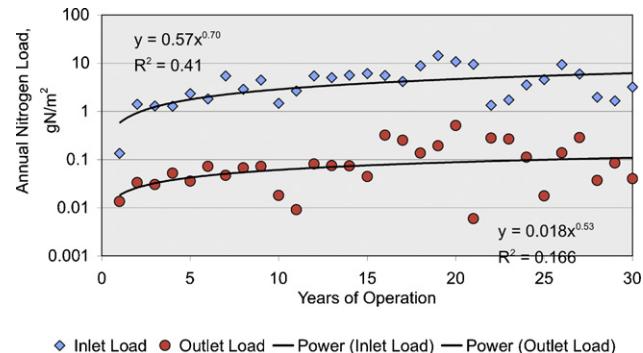
average of 5.41 gP/m<sup>2</sup> year was added in the pumped water, and 0.30 gP/m<sup>2</sup> year left the irrigation area.

Rates may also be viewed on an annualized basis, although additions did not occur over the entire year. The annualized rate of TP addition is the mass of phosphorus added per unit area in a given year, divided by the entire year length. This measure includes the period of zero flows as well as the pumping season. On this basis, 1.87 gP/m<sup>2</sup> year was pumped and 0.11 gP/m<sup>2</sup> year was exported from the irrigation zone. The annual loading of phosphorus to the wetland irrigation area increased slightly with time (Fig. 9). However, the annual phosphorus loading exported increased more substantially during the first 15 years, and continued to slowly increase over the last 15 years.

### 5.1.2. Nitrogen

The lagoon discharge added a total of 135.0 metric tons of dissolved inorganic nitrogen (DIN) to the wetland irrigation area over the 30-year period. A total of 3.4 metric tons were exported, thus creating a 97% mass removal. This calculation does not include atmospheric and organic sources, which are estimated to be very large (Table 3). The mean concentration reduction for DIN was 95%. On an instantaneous basis, an average of 12.46 gN/m<sup>2</sup> year was added in the pumped water, and 0.31 gN/m<sup>2</sup> year left in the irrigation area.

When rates are viewed on an annualized basis, 4.50 gN/m<sup>2</sup> year of DIN was pumped and 0.11 gN/m<sup>2</sup> year was exported from the irrigation zone. The annual loading

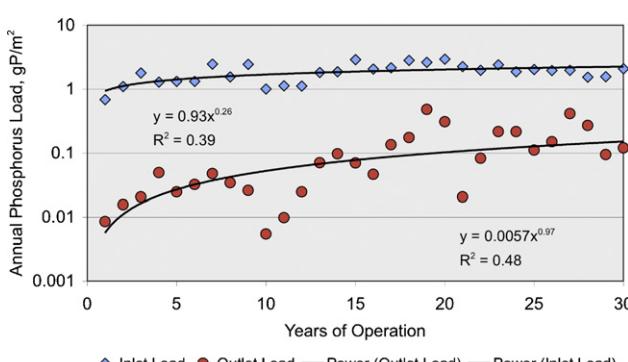
**Fig. 10 – Progressions of the annual DIN loadings into and out of the irrigation area.**

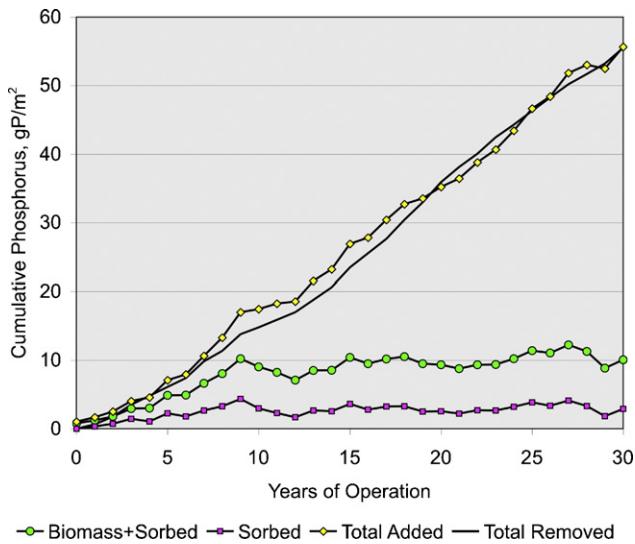
of DIN to the wetland irrigation area increased slightly with time (Fig. 10). The pumping loadings and the export loadings were closely parallel throughout the entire 30-year POR.

### 5.1.3. Storage of nutrients

The removed phosphorus was in part stored in new soil sorption, in part in increased plant and microbial biomass, and in part in new soil accretions. Phosphorus sorption was based on a 30-cm soil horizon and laboratory isotherms, plus water-column P concentrations. Measured biomass phosphorus content was used to calibrate an allocation model (Kadlec, 1997). The sum of the modeled storages was calibrated to be the same as the observed removal.

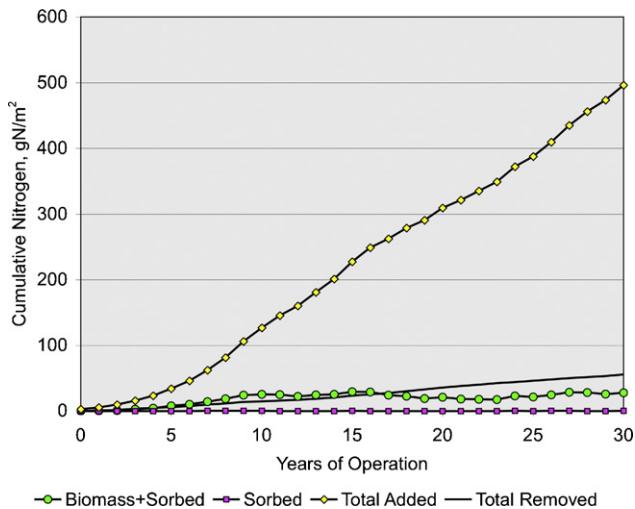
That model has been used to apportion stored phosphorus along the gradient in the irrigation area, as a function of time over the 30-year POR. Results of that modeling are presented in Kadlec (2009) and Kadlec and Bevis (2009); here a brief summary is presented to account for the removed phosphorus (Fig. 11). For the first 8 years, both sorption and biomass content increased. This is primarily due to the increase in the affected area. Thereafter, the sorbed and biomass pools of phosphorus remained relatively unchanged. Over the entire POR, accretion was the dominant storage mechanism. These model results are roughly independently confirmed by the observed soil and sediment accretion of about 15 cm on average (topographical surveys), at approximately 0.25% dry weight (dw) of phosphorus (laboratory core analyses) (Kadlec, 2009).

**Fig. 9 – Progressions of annual phosphorus loadings into and out of the irrigation area.**



**Fig. 11 – Allocation of phosphorus among storage compartments within the irrigation area, including sorption, biomass storage and accretion of new residuals. Sorption presumes 30 cm of affected soil depth, together with measured isotherms. Biomass nutrient content and accretion were modeled. The removal was computed from water inflow-outflow information.**

The storage of nitrogen was substantially different. Firstly, the quantities involved were much greater, as evidenced by the vertical scale in Fig. 12 being ten times that in Fig. 11. Secondly, the biomass model was calibrated to phosphorus removal, and consequently biomass nitrogen buildup and accretion of nitrogen were fixed by prior biomass calculations plus measured tissue and soil nitrogen concentrations. As for phosphorus,



**Fig. 12 – Allocation of nitrogen among storage compartments within the irrigation area, including sorption, biomass storage and accretion of new residuals. Sorption presumes 30 cm of affected soil depth, together with measured isotherms. Biomass nutrient content and accretion were modeled. The removal was computed from water inflow-outflow information.**

nitrogen sorption was based on a 30-cm soil horizon and laboratory isotherms, plus water column  $\text{NH}_4\text{N}$  concentrations.

The allocation of nitrogen strongly elucidates the fact that nitrogen removed from the flowing water was insufficient to account for the accretion of nitrogen. Both the observed and the modeled estimates show that there was ten times as much nitrogen in new soils and sediments as DIN removed from water (Fig. 12). One source of the discrepancy may have been due to organic nitrogen, which was not monitored for this project. However, the concentrations observed in similar lagoon effluents produce an estimate of 5 mg/L (mean of 5.08 mg/L for 76 lagoons; Price et al., 1995) entering the wetland; and wetland data would suggest about 1.5 mg/L leaving the irrigation area. If all of this organic nitrogen were consumed in plant growth, there still remains a large deficit. The required nitrogen presumably came from atmospheric nitrogen fixation (Kadlec and Bevis, 2009).

## 5.2. Reaction rate models

Input-output removals do not adequately characterize the performance of a wetland, because they do not represent internal phenomena. Strong internal gradients of water quality were observed in the irrigation area, as concentrations approached background values as water flowed downstream. Simple rules adequately describe the internal gradients, but these are different for phosphorus and nitrogen.

Removals are strongly tied to patterns of water movement. A general observation for wetland water flows is that they possess a distribution of detention times. Some water follows fast paths; other parcels of water follow slower trajectories. In constructed wetlands, it is often possible to use tracer testing to establish that distribution of detention times (Headley and Kadlec, 2007). In the case of the irrigation area of the Porter Ranch wetland, tracer testing was virtually impossible, because there was no defined outlet point at which to monitor for the tracer. Of necessity, it is here assumed that the irrigation area had a distribution of detention times characterized by four well-mixed units in series. This is the central tendency observed in a large number of wetland tracer tests (Kadlec and Wallace, 2008).

In general, the irrigation area experienced a 20% net water loss over the course of a pumping season. This causes two competing effects: the water slows as it travels allowing more time for reaction, and evaporative concentration occurs. These act in opposition, and it may be shown that the assumption of constant flow, at the inlet value, may be used without much error at this level of water loss (error less than 5% if water loss less than 20%) (see for example the methods and results of Kadlec and Wallace, 2008).

### 5.2.1. Phosphorus

Phosphorus is removed by uptake into biomass and the subsequent accretion of some small fraction of that uptake in the new soils and sediments that form from the resulting necromass (Kadlec, 1997). Accordingly, removals were higher near the discharge pipeline, where the high water concentrations caused high biomass. The biomass fertilizer response is taken

to be a Monod formulation:

$$N = N_{\max} \left( \frac{(C - C')}{(C - C') + s} \right) \quad (5)$$

where  $C$ =local P concentration ( $\text{gP/m}^3$ );  $C'$ =lowest P concentration that supports growth ( $\text{gP/m}^3$ );  $N$ =local biomass density ( $\text{g/m}^2$ );  $N_{\max}$ =maximum biomass density that can exist ( $\text{g/m}^2$ );  $s$ =half saturation P concentration for biomass ( $\text{gP/m}^3$ ).

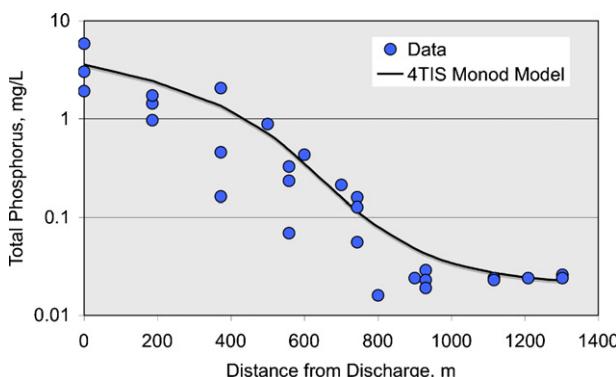
In the long-term sustainable state, neither sorption nor biomass expansion contributes to removal, but accretion was the dominant storage process. Biomass has completed its grow-in, and exists at the local maximum dictated by the water concentration (Eq. (5)). The biomass compartments are in a long-term mean, invariant condition; but with stochastic variations. In many situations, atmospheric processes are negligible. For the tanks-in-series (TIS) flow pattern, for tank 1:

$$q(C_0 - C_1) = k_N N = K \left( \frac{(C_1 - C')}{(C_1 - C') + s} \right) \quad (6)$$

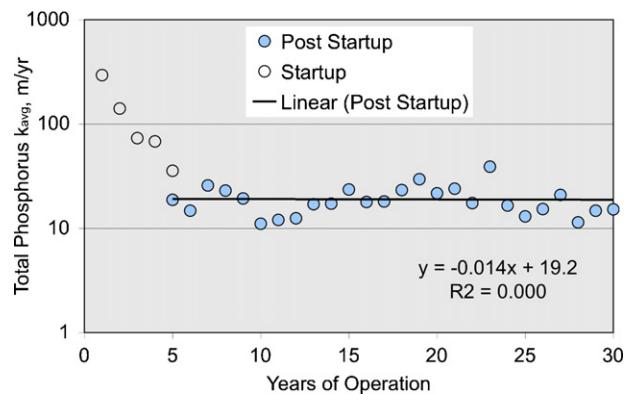
where  $C_0$ =local P concentration entering tank 1 ( $\text{gP/m}^3$ );  $C_1$ =local P concentration leaving tank 1 ( $\text{gP/m}^3$ );  $k_N$ =first order biomass removal rate coefficient ( $\text{gP/g/year}$ );  $K = k_N N_{\max}$ =autobiotic rate coefficient ( $\text{gP/m}^2 \text{year}$ );  $q$ =instantaneous hydraulic loading rate for tank 1 ( $\text{m/year}$ ).

Eq. (6) is solved sequentially for each of the four tanks, to produce the concentration profile through the irrigation area. The parameters  $K$  and  $s$  are adjusted to obtain a best fit of the observed data, via minimization of the sum of the squared errors. An example of such a fit is shown in Fig. 13 for the year 1996.

In effect, the biomachine model leads to a Monod removal rate expression in the fully developed state. It suggests zero order behavior at high P concentrations, and first order at low concentrations. At very high P concentrations, the zero order removal rate is given by  $K$ . However, the value of  $N$  at the inlet to the wetland is not the global maximum that could exist at still higher concentrations. The mean inlet concentration was  $3.4 \pm 1.0 \text{ mg/L}$ , yielding inlet values of  $N/N_{\max} = 0.94 \pm 0.02$ .



**Fig. 13 – Monod model fit for transect TP data for the 1996 pumping season. The parameters were  $K = 19.7 \text{ gP/m}^2 \text{ year}$  and  $s = 0.2 \text{ mg/L}$ ; the background concentration was  $0.02 \text{ mg/L}$ .**



**Fig. 14 – Time sequence of average rate coefficients for phosphorus removal in the irrigation area.**

Thus the biomass density achieved at the wetland inlet was about 94% of the maximum possible. According to this calibration, the removal was nearly zero order in the vicinity of the inlet, with a removal rate of  $13.5 \text{ gP/m}^2 \text{ year}$ , on an instantaneous basis, or  $4.7 \text{ gP/m}^2 \text{ year}$ , on an annualized basis.

A simple first order concentration-based model may also be used, in which the removal is given by

$$J = k(C_1 - C') \quad (7)$$

where  $J$ =P removal rate ( $\text{gP/m}^2 \text{ year}$ );  $k$ =first order removal rate coefficient ( $\text{m/year}$ ).

By comparison of Eqs. (6) and (7), the apparent first order areal, concentration-based rate coefficient is

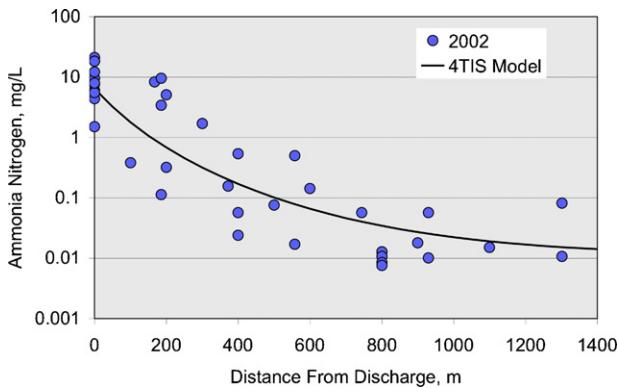
$$k = \frac{K}{(C_1 - C') + s} \quad (8)$$

At very low concentrations, where  $C_1 \approx C'$ , the first order areal rate coefficient  $k_{\max} = K/s$ . At higher concentrations, the apparent first order rate coefficient is dramatically reduced. During the stable period 1982–2007, the inlet zone rate coefficient was  $k = 4.1 \text{ m/year}$ , while the value was  $k_{\max} = 33.9 \text{ m/year}$  at the fringes of the irrigation zone.

The values  $s = 0.2 \text{ mg/L}$  and  $C' = 0.02 \text{ mg/L}$  were selected for data fitting for all years, while the values of  $K$  were adjusted for each year individually. The first few years of operation yielded high  $K$ -values, because grow-in and sorption were still actively augmenting the accretion process. During the stable period 1982–2007,  $K = 14.4 \pm 4.9 \text{ gP/m}^2 \text{ year}$ , which represents the maximum rate of removal at high concentrations. The first order areal rate coefficients varied from inlet to outlet. The average of inlet and outlet  $k$ -values had a 26-year mean of  $k_{\text{avg}} = 19 \pm 6 \text{ m/year}$ . Importantly, there were no time trends in any of the post-startup coefficients, as indicated for  $k_{\text{avg}}$  in Fig. 14.

### 5.2.2. Ammonia nitrogen

Nitrogen was in short supply within the wetland irrigation area, and the input with wastewater was a minor contribution to that supply. As a consequence, the utilization of incoming ammonia occurred against a background of other nitrogen sources for growth. As a result, the Monod formulation did not



**Fig. 15 – Areal model fit for transect NH<sub>4</sub>N data for the 2002 pumping season. The rate coefficient was  $k_A = 23.8 \text{ m/year}$  and the background concentration was 0.01 mg/L.**

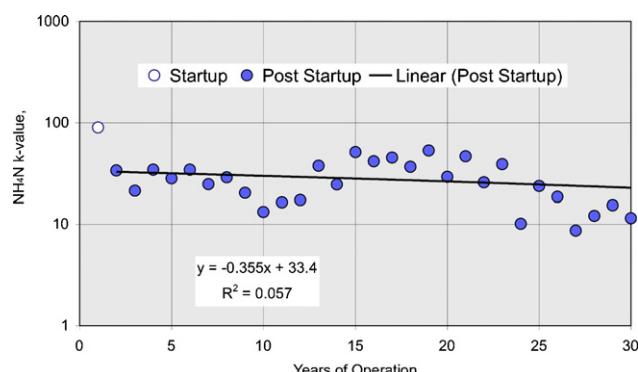
appear to be warranted, and a first order areal, concentration-based rate formulation was found to suffice. For each tank of four in series:

$$q(C_{A0} - C_{A1}) = k_A(C_{A1} - C'_A) \quad (9)$$

where  $C_{A0}$  = local NH<sub>4</sub>N concentration entering tank 1 (gN/m<sup>3</sup>);  $C_{A1}$  = local NH<sub>4</sub>N concentration leaving tank 1 (gN/m<sup>3</sup>);  $C'_A$  = lowest NH<sub>4</sub>N concentration (gN/m<sup>3</sup>);  $k_A$  = first order areal ammonia removal rate coefficient (m/year);  $q$  = instantaneous hydraulic loading rate for tank 1 (m/year).

Eq. (9) was solved sequentially for each of the four tanks, to produce the concentration profile through the irrigation area. The value  $C' = 0.01 \text{ mg/L}$  was selected for data fitting for all years. The parameter  $k_A$  was adjusted to obtain a best fit of the observed data for each year, via minimization of the sum of the squared errors. An example of such a fit is shown in Fig. 15 for the year 2002.

The first year of operation yielded a high  $k_A$ -value, probably because grow-in and sorption were still actively augmenting the removal process. During the stable period 1979–2007,  $k_A = 29.8 \pm 8.7 \text{ m/year}$ . Importantly, there was only a slight downward time trend in the ammonia rate coefficient, as indicated in Fig. 16.



**Fig. 16 – Time sequence of average rate coefficients for ammonia removal in the irrigation area.**

### 5.2.3. Nitrate nitrogen

Nitrate was measured in the lagoon discharge for 26 of the 30 years, and averaged  $1.62 \pm 1.44 \text{ mg/L}$  ( $N = 392$ ). Nitrate was studied along transects during 1987 and 1992 only, but it was routinely measured at the lagoon discharge and at the SW3X (SW31–SW34) stations (Fig. 3). Transects showed no spatial variability past the first 100 m from the discharge, with a background averaging 0.29 mg/L. This value was somewhat higher than the discharge from the enclosing wetland at stations E8 and E9 (ca. 0.15 mg/L, Table 2). At the SW3X stations, nitrate averaged  $0.16 \pm 0.31 \text{ mg/L}$  (mean  $\pm$  s.d.) ( $N = 528$ ). However, 35% of the samples at SW3X stations were below detection limits, and therefore one-half the detection limit was used in averaging. The indication from this data is that nitrate was quickly removed to background levels as the water moved through the irrigation area near the discharge pipeline.

However, higher values of nitrate were found episodically, and in some instances for protracted periods of time. For instance, from 1998 through 2004, the average concentration at SW3X stations was  $0.50 \pm 0.19 \text{ mg/L}$ . The reason for these excursions is not known.

The rapid disappearance of nitrate confirmed similar results from the pilot study (1976–77). That pilot project operated in a radial configuration, at a flow rate of 336 m<sup>3</sup>/d. In those studies, nitrate nitrogen was reduced from 1.50 mg/L to below detection (LDL = 0.10 mg/L) (Tilton and Kadlec, 1979). The corresponding nitrate rate coefficient was 138 m/year. In 1986, during the full-scale project POR, in situ field microcosms were dosed with nitrate in the field, and monitored for the disappearance of the nitrate. Bottomless 7-L buckets were twisted into the peat, and a nitrate solution was added. Disappearance followed an exponential decline with mean  $R^2 = 0.87$ . The mean of eight replicates gave a rate coefficient of 133 m/year. These smaller-scale precursor studies are consistent with the rapid disappearance of nitrate in the full-scale project, because they would forecast disappearance of nitrate quite near the discharge pipeline. They do not assist in explaining the observed background concentrations in the enclosing wetland discharge (Table 2), or at the edges of the irrigation area.

### 5.3. Ancillary water quality

Although nutrients were the primary design focus of the project, a number of other water quality parameters were also of interest. TSS, BOD<sub>5</sub>, DO, pH and pathogens were permit compliance parameters for some of the variants of the project permits. Redox potential and sulfur were briefly investigated. Metals were checked on a few occasions, and chloride and EC were used as tracers of the wastewater in the wetland.

#### 5.3.1. TSS

All versions of the NPDES permits required monitoring of TSS in the lagoon discharge, and for the last several years of the 30-year period, imposed an effluent limit of 30 mg/L. However, as discussed previously, TSS was very difficult to measure at internal wetland stations, and consequently TSS was not routinely monitored at any other locations. The overall POR TSS from the lagoon was  $33 \pm 22 \text{ mg/L}$ . However, for 1993 and prior, the average was  $17 \pm 4 \text{ mg/L}$ , and for 1994 and subsequent,

the average was  $50 \pm 20$  mg/L. As a consequence, there were numerous exceedences of the permit limit during the last 14 years.

Internal wetland TSS measurements were conducted during the 1981 operating season, on three dates and at ten distances from the discharge. At control locations,  $TSS = 7 \pm 3$  mg/L, while within the zone near the discharge,  $TSS = 34 \pm 28$  mg/L. During that season, the pumped  $TSS = 22 \pm 6$  mg/L. The higher TSS in the discharge zone may have been due to increased biological activity in this fertilized area. Background values were low, probably due to ample time for settling and very low linear water velocities.

A large proportion of the TSS in the lagoon was attributable to organic material, as evidenced by volatile suspended solids of approximately 70% of the total. The pumped water often had a “pea green” color, characteristic of high algal concentrations, especially in the last half of the POR. The chlorophyll content of the water was checked at several distances from the discharge on several dates during 1979, 1980, and 1984. The pumped water had variable chlorophyll-a, ranging from 75 to 780  $\mu\text{g}/\text{L}$ . Immediately near the discharge in the irrigation area, chlorophyll-a averaged 2500  $\mu\text{g}/\text{L}$ . This amplification was likely the result of settling and filtration of algae, together with growth in the wetland environment. At more remote points in the irrigation area, chlorophyll-a averaged 740  $\mu\text{g}/\text{L}$ . In control areas, beyond the influence of added nutrients chlorophyll-a averaged 200  $\mu\text{g}/\text{L}$ .

The dense cattail growth that developed near the discharge was undoubtedly inimical to algal growth, because of lack of light penetration. It may be presumed that there were no algae that survived underneath the floating mat after its development, because there was total darkness there.

### 5.3.2. BOD

Biochemical oxygen demand and  $\text{CBOD}_5$  were measured for the lagoon discharge to the wetland, for 23 and 9 years, respectively. All versions of the NPDES permits required monitoring of  $\text{BOD}_5$  (early years) or  $\text{CBOD}_5$  (later years) in the lagoon discharge, and for the last several years of the 30-year period, a limit of  $\text{BOD}_5 < 30$  mg/L was imposed on the lagoon discharges.  $\text{BOD}_5$  was routinely monitored at the enclosing wetland outlets (E8 and E9), but not at locations interior to the irrigation area. During 1981–1993,  $\text{BOD}_5$  from the lagoon was  $13 \pm 4$  mg/L. However, for 1994–2004, the average was  $32 \pm 18$  mg/L. As a consequence, there were numerous exceedences of the permit limit during the latter period. As of 1999, the measurement was switched to  $\text{CBOD}_5$ , which averaged  $14 \pm 4$  mg/L over the ensuing 12 years.

$\text{BOD}_5$  was measured at stations E8 and E9 over the period 1981–2000, and averaged  $3.4 \pm 2.7$  mg/L. As of 2001, the measurement was switched to  $\text{CBOD}_5$ , which averaged  $1.7 \pm 1.0$  mg/L over the ensuing 7 years.

As a check on the rate at which  $\text{BOD}_5$  was removed in the irrigation zone, a transect was run in August 1985. The pumped inlet  $\text{BOD}_5$  of 16 mg/L was reduced to a background of 5.0 mg/L within 400 m of the discharge.

### 5.3.3. Pathogens

Total coliform bacteria were monitored for the pre-project wetland in 1974. Total coliforms had a geomean of

334 cfu/100 mL for interior marsh stations ( $N=27$ ), and 226 cfu/100 mL for enclosing wetland inflows and outflows ( $N=21$ ). Enclosing wetland inflows and outflows had a geomean of 271 cfu/100 mL during the pilot project. The pilot project had a small irrigation area (ca. 5 ha), within which the total coliform count had a geomean of 249 cfu/100 mL. The pumped water during the pilot tests contained a geomean of 291 cfu/100 mL. Thus there were close to the same numbers of total coliforms in all waters during the pre-project and pilot years (1974–1977). Fecal coliform bacteria were present in the lagoon effluent at 99 cfu/100 mL for the pilot project, and were found in the pilot irrigation area at 31 cfu/100 mL.

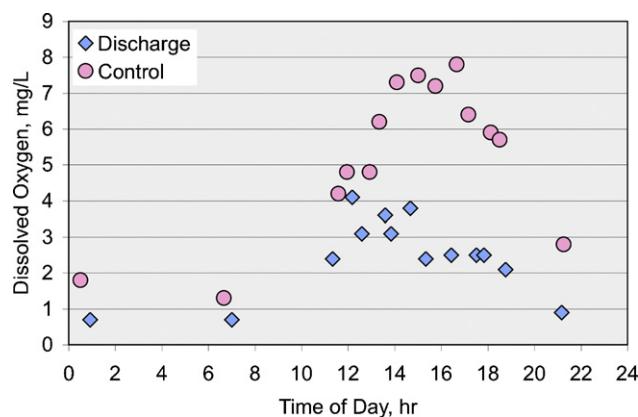
Fecal coliform bacteria were assayed in the lagoon effluent on a weekly basis during irrigation. The annual geomeans averaged 85 cfu/100 mL for the 30-year POR ( $N=474$ ). Fecal coliform bacteria were found in the full-scale irrigation area at 50 cfu/100 mL ( $N=34$ ), at distances of 100–200 m from the discharge pipeline during the startup years of 1978–1980. Because of the low numbers of fecal coliforms in the lagoon effluent, and the yet lower numbers found in the irrigation area, bacteriological monitoring of wetland stations was terminated after 1980.

Viruses were assayed during the pilot project in 1977. Although echovirus 32 was identified in the treatment lagoons, no virus was found in the holding pond discharge, within the pilot treatment zone, or in wetland controls.

The full-scale project was examined for virus during a synoptic investigation in 1978, and both reovirus and poliovirus were found at a number of locations (McDaniels, 1979). Quantification was via the cytopathogenic effect (CPE) in cultures of standard cell lines, which is measured as the infective dose that causes CPE in 50% of the inoculated plates ( $LD_{50}$ ). It is reported as the  $\log_{10}(LD_{50})$  per 0.1 mL, of a concentrate of specified strength. Here, the concentration factor is accounted, and the CPE units of the original water are reported. The aerated treatment lagoons contained 21 CPE units. The dechlorination pond contained 0.25 CPE units. The wetland at the discharge pipeline had 31 CPE units, while a control wetland site had 0.31 CPE units. Thus both the lagoons and the wetland were effective in reducing virus numbers, but not in their entirety.

### 5.3.4. DO and redox

Dissolved oxygen was measured in the lagoon discharge from 1999 to 2007, as a permit requirement. The average of weekly measurements was  $6.60 \pm 1.88$  mg/L ( $N=163$ ). The general effect of the lagoon effluent was to lower the oxygen status of the wetland in the vicinity of the discharge. The diurnal patterns of DO were typically lessened in the discharge area (Fig. 17). However, in areas of open water in the irrigation zone, where algal activity was high, the diurnal cycle was amplified. Such areas of open water were few in number, with most of the wetland covered by dense emergent vegetation. It should be noted that before such a shading canopy develops, the nutrients in the effluent fostered the growth of algae in the receiving wetland for a period of time. As a consequence of the oxygen production of these algae, very high DO values can be generated. In the 1976 pilot studies, DO in the receiving wetland reached about twice the saturation values, or about 20 mg/L, attributed to the high algal growth observed (Schwegler, 1978).

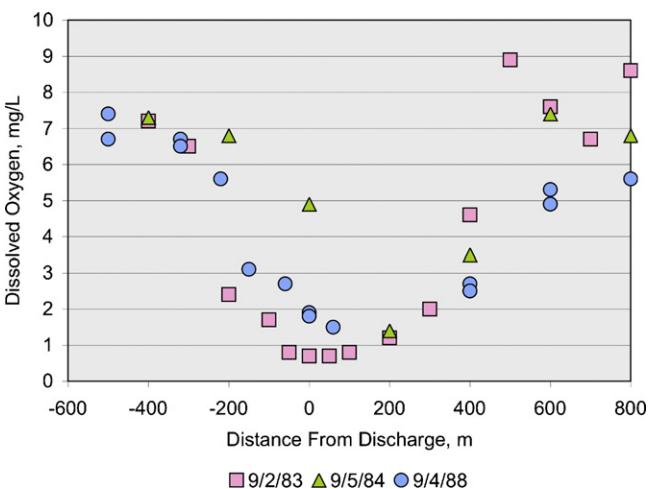


**Fig. 17 – Example diurnal patterns of DO in discharge and control areas for May 10, 1979.**

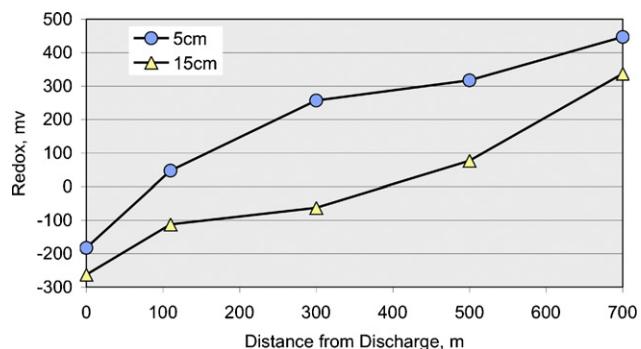
Dissolved oxygen was depleted in the discharge zone during the post-startup years of the full-scale project. It is likely that the incoming ammonia and  $BOD_5$  caused utilization of the incoming DO, creating low DO until the travel time caused reduction in those oxygen consumption processes. This gives rise to “oxygen sag” in the immediate vicinity of the discharge (Fig. 18). However, the quantification of utilization and resupply is not straight-forward (Kadlec and Knight, 1996; Kadlec and Wallace, 2008).

In the later years of the project, the floating mat would have precluded reaeration, and total darkness in the water column would have extinguished algal growth.

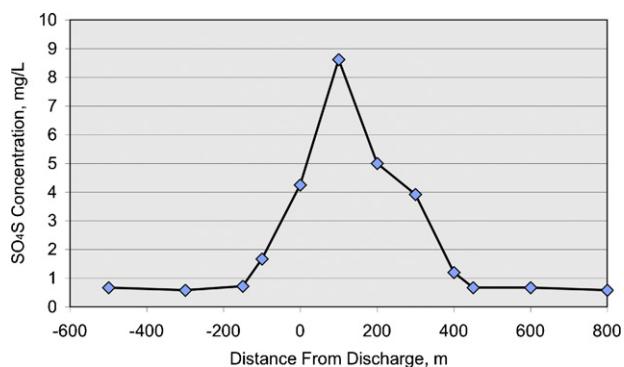
The redox potential in wetland soils is known to mediate the pollutant removal processes that occur there. In the overlying water, redox measurements can extend the scale of potential to values in the reducing range, and thus provide an alternate to dissolved oxygen measurements. Fig. 19 shows the results of a synoptic set of redox potential measurements along the gradient away from the discharge in the irrigation



**Fig. 18 – Examples of oxygen sag in the discharge zone.** Distance in this chart has been retained as actual upgradient and downgradient measures, to illustrate the recovery of DO in both directions. Values are from mid-day measurements.



**Fig. 19 – Redox potential at two depths in the water column on 9/3/89.**



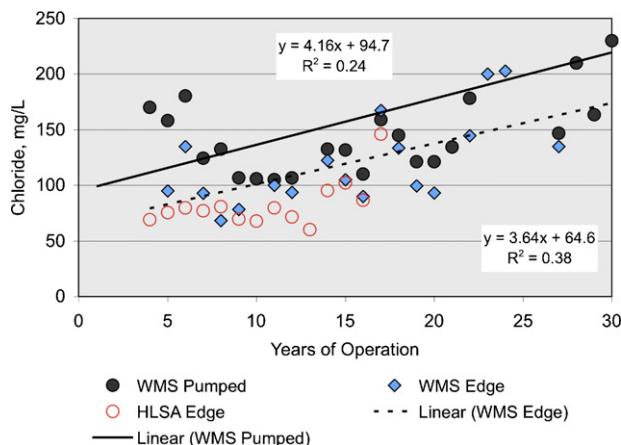
**Fig. 20 – Sulfate sulfur in the discharge zone.** Distance in this chart has been retained as actual upgradient and downgradient measures, to illustrate the dissipation of  $SO_4S$  in both directions. Values are means of measurements on two dates.

area. In close proximity to the discharge, the water column contained no oxygen, and further, the redox potentials were very low in the water column. The water has been driven to anaerobic conditions at the soil–water interface, within the litter layer.

### 5.3.5. Sulfate and sulfur

Sulfate sulfur ( $SO_4S$ ) was monitored in 1985 at a number of locations. Sulfate was present at low levels in the lagoon discharge to the wetland (2.6 mgS/L), and in the rainfall impinging on the wetland (1.6 mgS/L). At control locations in the enclosing wetland,  $SO_4S = 0.65$  mgS/L, and the outflow at station E8 was at 0.63 mgS/L. However, levels were higher in the irrigation area, with a spatial mean of 2.5 mgS/L, but an internal peak of 8.6 mgS/L (Fig. 20). The extent of the elevated  $SO_4S$  was entirely within the irrigation area.

A mass balance for the 1985 season showed 982 kgS entering the irrigation area with lagoon water, and 546 kgS entering with rainfall (Kadlec and Alvord, 1989). Outputs were 225 kgS leaving in flowing water, and 193 kgS stored in a deeper pool of wetland water at the end of the irrigation season. The balance of 1111 kgS was removed in the wetland irrigation area. This removed sulfur could be found in the newly accreted solids, according to solids content measurements. Interestingly, at least some small portion of the sulfur was identified to be in the form of elemental sulfur.



**Fig. 21 – Chloride entering and leaving the irrigation area.**

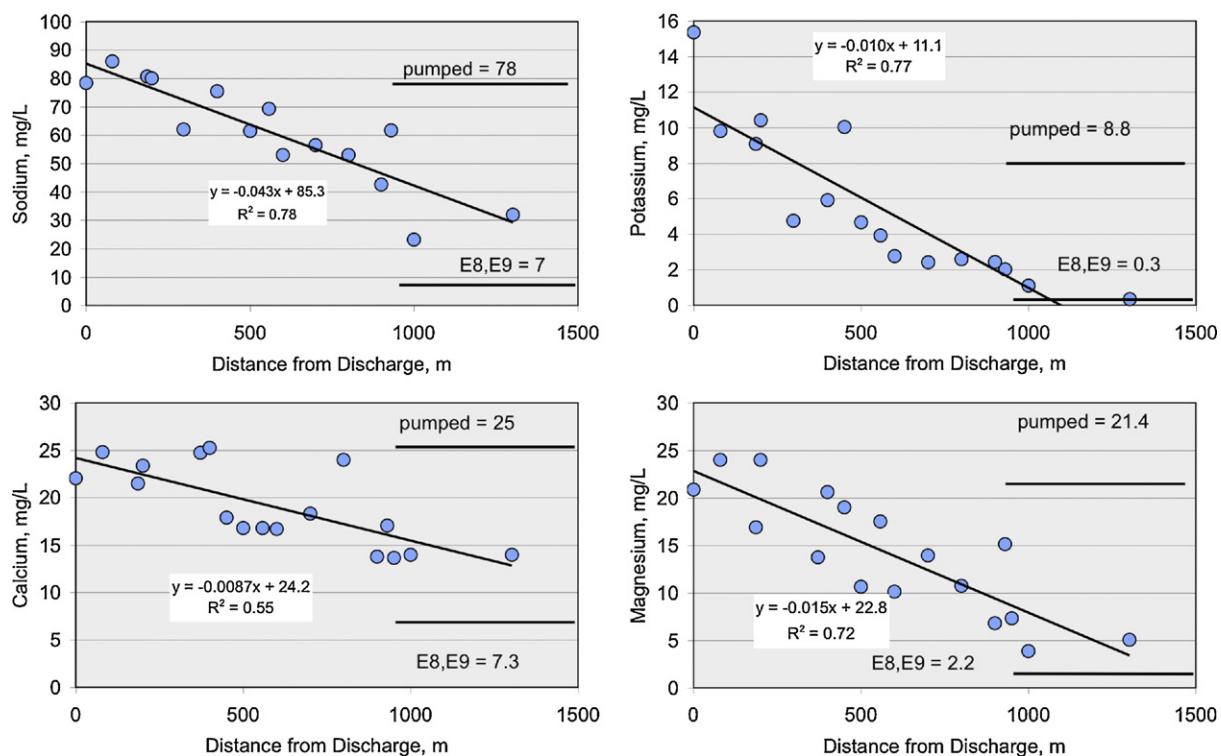
### 5.3.6. Chloride, EC, pH

Chloride entering the irrigation area had an annual average of  $123 \pm 36 \text{ mg/L}$ , and at the outflow line (SW3X line) was  $100 \pm 32 \text{ mg/L}$  for the 25 years for which this parameter was measured by HLSA. However, the annual averages of samples collected and analyzed by WMS for the pumped water and at the edges of the 100-ha irrigation zone were  $155 \pm 49$  and  $120 \pm 39 \text{ mg/L}$  respectively. Thus there was a moderate loss of this nearly conservative element in the irrigation zone, of about 20%. These concentrations were well above the wetland background level of about  $28 \text{ mg/L}$  (Table 1). There were steep time trends in chloride entering the wetland, and in the outflows from the irrigation area (Fig. 21). The rates of increase were  $4.4\% \text{ year}^{-1}$  and  $5.6\% \text{ year}^{-1}$ , respectively, and hence both

nearly doubled over the POR. Data from 94 transects suggest a gradual decline with distance from the discharge pipe, in the direction of water flow, to the edge of the irrigation zone ( $y = 159 - 0.06x$ ,  $R^2 = 0.10$ ).

Electrical conductivity in the pre-project wetland was  $153 \mu\text{S}/\text{cm}$  (Table 1). EC in the pumped water was measured in 26 of 30 years, and the annuals average =  $728 \pm 161 \mu\text{S}/\text{cm}$ . EC in the outflow from the enclosing wetland (E8 and E9) was measured in 29 of 30 years, and the annuals average =  $117 \pm 46 \mu\text{S}/\text{cm}$ . This drop in EC occurred in large measure in the irrigation zone. The annuals average of samples collected and analyzed by WMS at the edges of the 100-ha irrigation zone over 18 years was  $486 \pm 187 \mu\text{S}/\text{cm}$ . There were strong time increase trends, of  $3.8\% \text{ year}^{-1}$  for the inflow, from  $500$  to  $900 \mu\text{S}/\text{cm}$ ; and  $8.5\% \text{ year}^{-1}$  for the outflow, from  $200$  to  $700 \mu\text{S}/\text{cm}$  leaving the irrigation area. The transect data show this decline was gradual from the discharge pipe, in the direction of water flow, to the edge of the irrigation zone ( $y = 723 - 0.36x$ ,  $R^2 = 0.28$ ).

The background pH in the pre-project marsh surface waters was  $6.38$  (Table 1), while the pumped water had an annuals average =  $8.02$  (Table 2). pH in the outflow from the enclosing wetland (E8 and E9) was measured in 22 of 30 years, with an annuals average =  $6.76 \pm 0.55$ . In contrast to the gradual time trends for chloride and EC, pH displayed a sudden increase at the irrigation area outflow, occurring in 1992–1994. During the early period of years 15–17, the pH at the irrigation area outflow had an annuals average of  $6.49$ . pH was not measured there in years 1–14. For years 16–30, the outflow annuals average was  $7.31$ . This was apparently not a direct response to the very slight increase in incoming pH of  $0.2\% \text{ year}^{-1}$ , which was from  $7.8$  to  $8.3$  over the 30-year POR. The transect data show the decline with travel across the irrigation area was gradual,



**Fig. 22 – Declines of common cations with distance from the discharge.**

in the direction of water flow, to the edge of the irrigation zone ( $y = 7.30 - 0.0005x$ ,  $R^2 = 0.06$ ).

### 5.3.7. Metals

Metals may be considered in two categories: those cations that are indicative of ecosystem status and found in relative abundance (Fe, Ca, Mg, Na and K), and the trace metals that may originate from sources within the community and find their way into the wetland via the community wastewater.

Two dozen rare earth elements were sought in water samples from the lagoon and wetland background stations in 1978. None were above detection limits of the neutron activation analytical procedure.

Iron is known to participate in the sorption of phosphorus, and is therefore of interest in connection with P removal. The pre-project wetland showed little dissolved iron in the surface water (0.20 mg/L, Table 1), but porewater averaged about ten times higher. Iron was similarly low in the outlet flows from the enclosing wetland, 0.16–0.19 mg/L. Iron was low in the pumped water, at about 0.11 mg/L. Transects in the irrigation area showed  $0.36 \pm 0.25$  mg/L in 1989. There were no trends in Fe concentration with distance from the discharge.

The hardness cations, Ca and Mg, display strong trends with distance in the irrigation area (Fig. 22). Magnesium was present in the pumped water at concentrations considerably greater than those in the pre-project wetland or its outflows. Within the irrigation zone, concentrations dropped rapidly, and were at or near background at the edge of the irrigation zone. Calcium was also pumped at substantially higher concentrations than background, and the water had reached only about half background at the edge of the irrigation zone.

Sodium was pumped at concentrations about ten times higher than in the pre-project wetland or its outflows (Fig. 22). Recovery to background was not complete at the outlet of the irrigation zone. Potassium, a macro-nutrient, responded the fastest, with the disappearance of all of the excess incoming K inside the irrigation zone.

### 5.4. Water temperature

Air and wetland water temperatures are subject to both diel and annual cycles, corresponding to the cycles in solar radiation (Kadlec, 1999). The swing in wetland water temperature from day to night was considerable. Hourly temperature logging was conducted at transect locations ranging from control to discharge in 1997 and 1998. The difference between the daily maxima and minima, over 144 of 153 season days, was  $6.8 \pm 3.3$  °C. The mean maximum daytime temperature of the wetland surface water was 20.8 °C, while the mean minimum was 14.0 °C.

The water temperatures varied seasonally. Data from the irrigation area and a control area were regressed to a truncated, sinusoidal time series model of the form:

$$T = T_{\text{avg}}(1 + A \cdot \cos[\omega(t - t_{\text{max}})]) \quad (10)$$

where  $A$  = fractional half amplitude of the annual temperature (°C);  $t$  = time (Julian day);  $t_{\text{max}}$  = time of annual maximum temperature (Julian day);  $T$  = water temperature (°C);  $T_{\text{avg}}$  = annual

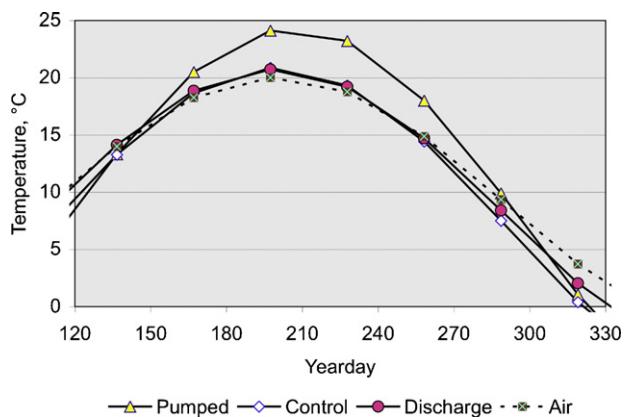


Fig. 23 – Water and air temperatures for 1998. Points are monthly averages of daily data, smoothed with a cosine trend.

average unfrozen water temperature (°C);  $\omega$  = annual frequency,  $\omega = 2\pi/365 = 0.0172$  year $^{-1}$ .

The various data fits showed the seasonal maximum occurred on July 19 (Julian day  $200 \pm 0.6$ ). That maximum was  $19.8 \pm 1.2$  °C, and the seasonal average water temperature along the transect was  $16.6 \pm 0.8$  °C. That was cooler than the pumped water, which averaged 19.9 °C for the season. The full seasonal time trend of air and water temperatures is shown in Fig. 23.

The adaptation of the warm incoming water to wetland energy balance conditions was rapid. Surface water transect temperature data show that the accommodation is complete in about 100 m from the discharge (Fig. 24).

### 5.5. Winter and off-season conditions

Because the discharges were made only during the unfrozen season (6 months), questions logically arise about events and processes during the remainder of the year. Therefore, samples and studies were conducted during the off-season,

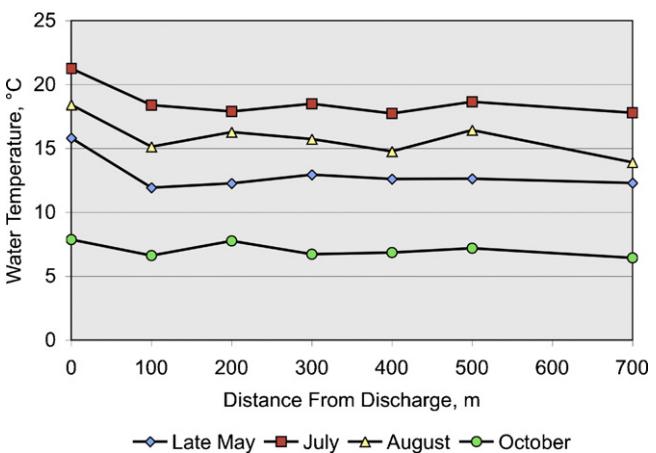


Fig. 24 – Longitudinal profiles of water temperature for the 5-year period 1998–2002, as a function of season. Note that the drop to the balance temperature occurred in the first hundred meters downstream of the discharge.

**Table 4 – Concentrations of chloride and EC in water and overlying ice. The distribution coefficient is the ratio of the ice concentration to the water concentration. Concentrations are means of triplicates.**

	Chloride			EC		
	Water (mg/L)	Ice (mg/L)	Distribution coefficient	Water ( $\mu\text{S}/\text{cm}$ )	Ice ( $\mu\text{S}/\text{cm}$ )	Distribution coefficient
<b>Control</b>						
2/2/84	157	40	0.25	435	53	0.12
11/7/84	101	7	0.07	105	20	0.19
1/14/85	106	4	0.04	101	4	0.04
Mean			0.12			0.12
<b>Discharge</b>						
2/2/84	128	58	0.45	409	88	0.22
11/7/84	102	11	0.11	142	24	0.17
1/14/85	90	18	0.20	87	23	0.27
Mean			0.25			0.22

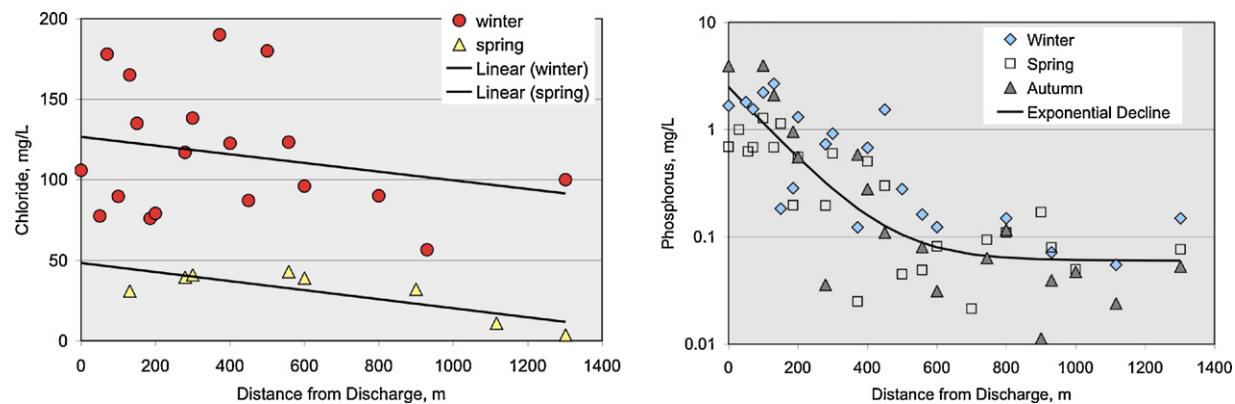
although the frequency and intensity was much less than during operation.

The growing season in this region ends during early September, and senescence of above-ground plant parts is generally complete by the end of that month. October is a month of generally ice-free conditions, but with zero emergent growth. Ice formation typically begins about November 1 each year, as does the accumulation of a snow blanket. Depending upon the sequence of snow and cold, ice thicknesses varied considerably. If a thin layer of ice formed first, and was followed by large snow accumulation, there were essentially ice-free conditions beneath that snow. A thickness of half a meter of snow essentially prevented further ice formation. Conversely, if cold conditions were not accompanied by significant snowfall, the wetland surface waters and top layers of exposed soils would freeze. Frost depths were not observed to be greater than 20 cm in any year, but that typically included the entire surface water sheet that existed at the time of freeze-up. Frost that formed in peat soils, especially in the shrub cover type, was persistent in some years. Frozen peat was observed throughout the entire month of May in some years, in the shrub cover types. In the same year, growth commenced at the end of April in the sedge and cattail cover types.

These conditions had several consequences for water quality during the off-season. Firstly, under-ice flow was essentially non-existent everywhere in the irrigation area,

because of the slight tilt to the wetland surface, the very shallow water depths, and the extremely low hydraulic conductivity of the peat soils. In other words, the irrigation area became capped with ice, and went to a no-flow condition. However, flows from the enclosing wetland did not necessarily stop during winter, because of the relatively deep waters in the Deadhorse impoundment and beaver ponds near the outlets.

At the time of freeze-up, the surface waters of the irrigation zone contained various contaminants at concentrations that varied along the gradient. Those contaminants were typically excluded from the ice as it formed, and hence were driven vertically downward into under-ice water or porewater. Samples taken under the ice were therefore representative of the gradient from which the ice formed, for both nutrients and other substances. The ice has considerably lower concentrations. The relative lack of biological activity presumably prevented dissipation of nutrients, while more conservative substances, such as chloride and EC, were merely held in place by the ice cap. Information from three sampling dates indicates that the ratio of ice to water concentration is approximately 0.12 in control areas, and approximately 0.23 in the discharge area adjacent to the pipeline (Table 4). In those locations where the ice penetrates down to the peat, the excluded solutes are driven into the soil, and reside there in porewater for the winter. It should be noted that the partitioning of nitrogen and phosphorus is not as great as for chloride and EC. Further dis-

**Fig. 25 – Distributions of chloride and phosphorus along the flow direction during the off-season.**

cussion of this phenomenon for the Porter Ranch wetland may be found in Kadlec (1984), Kadlec et al. (1987b), Li (1985) and Kadlec and Li (1990).

The winter snowpack resides on top of the frozen wetland, and is typically isolated from the porewaters below. The spring melt proceeds from the top downward, and hence the melt water is of good quality, because both snow and spring rain quality were better than the lagoon effluent. This may be seen in transect data from early spring, which are of relatively good quality in terms of chloride (Fig. 25). However, because the solute exclusion from ice for nitrogen and phosphorus is not efficient, the spring melt simply restores the previous year's gradients of nutrients, perhaps with a slight dilution factor due to snow (Fig. 25).

## 6. Permits

The full-scale project operated under two categories of regulatory permits: a NPDES permit issued by the Michigan Department of Environmental Quality (MDEQ), and a special land use permit issued by the MDNR. The NPDES permit was applied for prior to the project startup in 1978, and issued approximately 10 years later. The first version deemed that an area interior to the enclosing wetland would serve as a mixing zone, and placed limits at stations located at the outlet of the irrigation zone. These limits reflected a desire for anti-degradation at points in the wetland further downstream. For instance, the target values for nutrients were  $TDP < 0.5 \text{ mg/L}$  and  $\text{NH}_4\text{N} < 3.0 \text{ mg/L}$ . The NPDES further specified monitoring of the outflows from the enclosing wetland, as a means of keeping track of the quality of discharges to the receiving streams. In 1998, the NPDES permit was revised to drop all requirements for monitoring inside and out of the wetland. Instead, the compliance point was moved to the discharge from the dechlorination pond, where secondary water quality limits were imposed.

The secondary limit for suspended solids,  $30 \text{ mg/L TSS}$  monthly maximum, proved difficult to meet for the lagoon system, in large measure because of algal particulate matter. The HLSA took corrective actions, including application of barley straw, and installation of surface aerators to supplement the existing perforated tube aeration. The numerous exceedences were deemed to be a violation, and large fines were levied on the HLSA. This situation was remedied by installation of sand filters after the dechlorination ponds, which were to be completed in 2008.

The Porter Ranch is managed by MDNR, under the direction of the USFWS. The property was purchased with Federal Funds, with the condition that designated uses would include research and public access. Accordingly, MDNR allowed the lagoon discharge to the wetland on condition that the hydrological and ecological effects be studied, separately from NPDES compliance monitoring, and at the expense of the HLSA. A special use permit was therefore in effect throughout the first 30 years of the project. That permit required annual reporting, and research proposals discussed and approved by MDNR each year. The terms of this permit were fully complied with. After year 30, MDNR has terminated this research, on the basis that it was irrelevant to their mission. Instead, the HLSA

is being charged an annual use fee, which is roughly double the annual cost of the previous research activities.

## 7. Economics

The discharge to the wetland was originally one of several options considered for treating and disposing of the lagoon effluent. For instance, the lagoon effluent could have been further treated to the tertiary level, including phosphorus removal, and piped to Bear Creek, a tributary of the Muskegon River, approximately 4 km from the lagoons. Tertiary treatment consisted of biological, chemical and physical components. Land application was also considered. The present worth of tertiary treatment was 2.8 times greater than the wetland option, and the present worth of land application was 1.7 times greater (Williams and Works, 1976). The facility life was taken to be 20 years in these preliminary estimates, but the system lasted 30 years with some repairs to wooden components. This original life estimate was based upon a configuration of the wetland containing impoundments, which did not materialize. Instead, the gated distribution pipe and boardwalk support were built. The actual construction cost was not far different from the Facility Plan estimate, however. In 2008 dollars, the wetland system components cost about 1.2 million dollars (Table 5). The wetland irrigation system (boardwalk and pipeline) cost about \$328,000 of that total. This capital cost was much lower than that for comparable sized constructed treatment wetlands (Kadlec and Wallace, 2008).

In 2008, the irrigation system was replaced. The irrigation pipe was found to be reusable, but the original wooden boardwalk and steel support posts were removed. The cost of the replacement irrigation system, including removal of the old system, was \$516,000. The transfer pipeline, from the WWTP to the wetland was not replaced, nor was the transfer pump.

The operations and maintenance (O&M) cost estimate for the wetland portion of the project was \$55,000 (2008 dollars), about half of which was for pumping energy. That pumping energy cost was in fact much lower than the estimate. The original plan called for water quality analyses to be performed at the WWTP, in a laboratory built as part of the larger lagoon

**Table 5 – Capital and estimated operations and maintenance costs for the Houghton Lake wetland treatment system (2008 dollars, ENR CCI = 8094)**

Capital costs	
Holding pond modification	\$112,546
Dechlorination pond	\$446,686
Pond-wetland water transfer	\$243,753
Irrigation system	\$328,892
Monitoring equipment	\$28,282
	\$1,160,159
Annual operations and maintenance	
Pumping	\$5,000
Monitoring	\$10,084
Maintenance	\$4,200
Research	\$24,000
	\$43,284

construction project. However, for much of the project history, samples were analyzed at a commercial analytical lab instead. Additionally, the original estimate did not include the cost of the ecological research required by the land use permit. In total, the actual O&M cost was approximately \$43,000 per year (2008 dollars) (Table 5). If prorated over the 100-ha irrigation area, this cost is at the lowest 20th percentile of the distribution of O&M costs for treatment wetlands (Kadlec and Wallace, 2008). However, the imposition of a special land use fee as a replacement for research has approximately doubled the future O&M cost, from 2008 forward.

## 8. Discussion

### 8.1. Phosphorus loading

The phosphorus loading graph is a useful tool with which to compare the performance of the Houghton Lake wetland with other wetlands. This plot of outlet concentration versus inlet loading has been described in several sources, including Kadlec and Wallace (2008). The P loading is simply the product of the incoming P concentration and the instantaneous hydraulic loading. That hydraulic loading is the amount of water applied divided by the wetland area in question. It represents the amount of P applied during the pumping season. The area affected by the discharge was determined from aerial photography in 24 of the 30 years, from changes in color and color infrared reflectance. Those changes were ground-truthed to be indicative of replacement communities or greatly increased vegetation density. That area increased in an exponential manner, according to

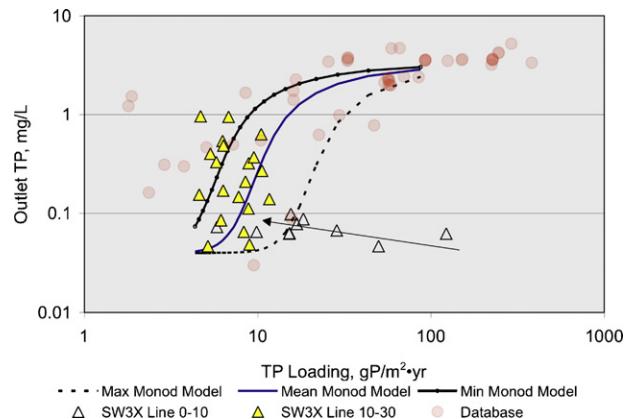
$$A = A_{\max} \left( 1 - \exp \left( -\frac{t}{\tau} \right) \right) \quad (11)$$

where  $A$  = affected area (ha);  $A_{\max}$  = maximum affected area (ha);  $t$  = time (years);  $\tau$  = time constant (years).

This relationship is described in more detail in Kadlec and Bevis (2009). The best-fit parameters were  $A_{\max} = 82.3$  ha and  $\tau = 9.0$  years, which produced a least squares fit with  $R^2 = 0.913$ . Areas in excess of this received essentially no P loading, and hence the loading was calculated using only the affected area. The concentrations for the loading plot are those measured at the edge of the 100-ha irrigation zone, at the stations along the SW3X line.

During the first 10 years of operation, the loading decreased because the treatment area (affected area) was rapidly expanding. However, P removal remained essentially constant, down to just above background (Fig. 26). In the last 20 years, the P removal extended out to and in some cases beyond the SW3X line. Thereafter, outlet P concentrations were erratic, with no particular time trend.

This phenomenon is better understood from the modeling perspective. The "K" parameter of the Monod model was variable from year to year, as were the hydraulic loadings to the irrigation area and the inlet concentrations (see for example Fig. 14). Fig. 26 plots the model results for the mean hydraulic loading (6.45 m/year instantaneous) and a half saturation constant of 0.2 mg/L; and the minimum, mean and maximum  $K = 8.6, 14.4$  and  $30.1 \text{ g/m}^2 \text{ year}$ . It should be noted that the



**Fig. 26 – Loading response for phosphorus over the POR.** The open triangles represent the first 10 years, and the arrow describes the trajectory. The closed triangles represent the last 20 years, and there is no particular sequence. The circles represent the performance of 50 other treatment wetlands, each for its entire POR. The lines are the Monod model for max, min and mean values of K-value (30.1, 8.6, 14.4 g/m<sup>2</sup> year), and the mean hydraulic loading (6.45 m/year), and half-saturation constant ( $s = 0.2 \text{ mg/L}$ ).

model calibrations for the several years were derived from transect data, and not the SW3X data shown on this plot. The performances for years 11–30 are seen to be bracketed by the model curves, which are widely separated for the range of P loadings that were experienced. Furthermore, the model curves are quite steep in that range of loadings. This is an indication of the extreme sensitivity of outlet concentrations to wetland conditions. Basically, there is a steep front of P concentrations that easily advances and recedes depending on conditions of particular year.

The performance of the Houghton Lake wetland was in some years the same as for comparison wetlands, and in some years better (Fig. 26), as indicated by lower outlet concentrations. This is likely due in large part to operation only during the growing season.

### 8.2. Apparent nitrogen deficit

Phosphorus was removed to storage in the form of new biomass and accretion of decomposition residuals. Those same materials also contained nitrogen as part of their structural makeup. The ratio of tissue nitrogen to tissue phosphorus ranged from 8 to 12 for the standing stocks of aboveground biomass, including live, standing dead and litter (Kadlec, 2009). The ratio was higher for the top 30 cm of soils, roots and rhizomes, ranging from 13 to 23. As a consequence, the accretion and burial of the removed phosphorus must of necessity have been accompanied by the accretion and burial of a good deal more nitrogen. Since most of the storage was in new soils and sediments, at least 10 times more nitrogen was accreted than phosphorus. However, the removal of nitrogen from the water was only 2.5 times the removal of phosphorus.

The average annual loading of DIN to the irrigation zone was  $4.50 \text{ gN/m}^2 \text{ year}$  (Fig. 10), and only  $0.11 \text{ gN/m}^2 \text{ year}$  left the irrigation zone, for an apparent average annual removal of  $4.39 \text{ gN/m}^2 \text{ year}$ . The average annual loading of phosphorus to the irrigation zone was  $1.87 \text{ gP/m}^2 \text{ year}$  (Fig. 9), and only  $0.11 \text{ gP/m}^2 \text{ year}$  left the irrigation zone, for an apparent average annual removal of  $1.76 \text{ gP/m}^2 \text{ year}$ . The removal of this amount of P should have been accompanied by not less than  $18.8 \text{ gN/m}^2 \text{ year}$  of nitrogen, of which only 4.39 could possibly have come from the added water, leaving a deficit of  $14.4 \text{ gN/m}^2 \text{ year}$ . However, it seems likely that some or all of the nitrate fraction of the DIN was lost to denitrification, and therefore the deficit was even larger. Approximately 20% of the incoming DIN was in the form of nitrate nitrogen (Table 2).

It is useful to speculate on the source of the extra nitrogen. While it may have been possible that some nitrogen was extracted from antecedent soils and re-deposited in the newly formed soils in the top layer, that source was not sustainable. Apart from the finite reserve that could have been depleted, the root zone moved upward over time and ultimately occupied a horizon that could no longer reach antecedent soils.

Atmospheric deposition surely accounted for some added DIN. Rainfall in the Houghton Lake area was assayed for nitrogen, and found to contain approximately  $0.6 \text{ mg/L}$  of DIN (Pecor et al., 1973). This source could have contributed about  $0.22 \text{ gN/m}^2 \text{ year}$  of nitrogen, which is a tiny portion of the requisite added nitrogen.

Biological nitrogen fixation is the process by which nitrogen gas in the atmosphere diffuses into solution and is reduced to ammonia nitrogen by autotrophic and heterotrophic bacteria, blue-green algae, and higher plants. Under anaerobic conditions, microbial assemblages in the root zone of *Typha* spp. were shown to fix considerable quantities of atmospheric nitrogen (Bristow, 1974). The anticipated range of annualized rates is approximately  $8\text{--}20 \text{ gN/m}^2 \text{ year}$ , based upon Waughman and Bellamy (1980). This range nicely brackets the observed nitrogen deficit of  $14.4 \text{ gN/m}^2 \text{ year}$ .

### 8.3. Oxygen sag and reaeration

The protection of receiving waters is very often promoted by regulation of the DO content of the discharged water. The standard is often a lower bound of  $5 \text{ mg/L}$  of DO, which is intended to be protective of freshwater aquatic life. The DO leaving the holding pond and entering the wetland was  $6.6 \pm 1.9 \text{ mg/L}$ . Excursions below the  $5 \text{ mg/L}$  level occurred for 13% of the weekly measurements. The accompanying CBOD<sub>5</sub> levels accompanying were  $14 \pm 10 \text{ mg/L}$ . Based on 70 dual measurements of BOD<sub>5</sub> and CBOD<sub>5</sub>, the corresponding BOD<sub>5</sub> levels were  $32 \text{ mg/L}$ . The background BOD<sub>5</sub> was approximately  $5 \text{ mg/L}$ . Therefore, reaeration would be required to dissipate the remaining excess of  $27 \text{ mg/L}$ , while maintaining the wetland background DO of about  $7 \text{ mg/L}$  (Fig. 18).

The hydraulic loading was approximately  $600,000 \text{ m}^3/\text{year}$ , applied during the pumping season, which averaged 128 days in length during the early years (Fig. 18). If the zone of oxygen sag is presumed to be  $64 \text{ ha}$ , then the required oxygen would be  $0.20 \text{ gO}/\text{m}^2 \text{ d}$  (Fig. 18). The re-aeration potential for other wetlands has been found to be about  $2.0 \text{ gO}/\text{m}^2 \text{ d}$  (Kadlec and Wallace, 2008), and therefore oxygen is likely being consumed

by processes other than BOD<sub>5</sub> reduction (carbon compounds and ammonia in the water). The primary candidate for the extra oxygen demand is the oxidation of carbonaceous materials in the wetland sediments (sediment oxygen demand, or SOD). Because the carbon cycle is very large in the irrigation zone, large amounts of decomposable material are produced there. Oxidation of that biomass appears to claim more oxygen than does reduction of BOD<sub>5</sub> in the incoming water.

### 8.4. Partial year operation

This project opted for winter storage of the treated water from the aerated lagoons. The primary reason for this seasonal discharge was that all treatment mechanisms were deemed to operate solely, or at least more optimally, in the warm season. Those options originally included infiltration to groundwater and under-drained infiltration to a receiving stream, as well as the wetland treatment option. The groundwater discharge option was abandoned at the start of wetland utilization, because of the incompatibility of the lagoon treatment with the groundwater permit. The under-drained infiltration fields were retained, and received a small amount of lagoon water for irrigation of a hay crop. The under-drain water was subject to a permit separate from that for the wetland. Therefore, the 12-ha storage pond was part of the original design of the treatment system. The dechlorination pond was added so that no chlorine residual would reach the wetland and impair its functions. However, chlorination was not used in the POR.

Constructed wetlands are capable of year-round function at comparable latitudes, under comparable climatic conditions (Herskowitz, 1986; Hanna et al., 2001; Phillips and Pries, 2001; Kadlec, 2008). Operation is hydraulically possible, despite freezing temperatures. Ice formation is variable in thickness, ranging from zero for early snow, to  $10\text{--}20 \text{ cm}$  for less insulation (Kadlec and Wallace, 2008). But, for many substances, wetland removal rates are slowed under cold temperature conditions. Therefore, the choice is often between storage or a larger wetland. For the Houghton Lake system, the existence of the storage pond rendered this a moot decision.

Given the summer loading of contaminants into the wetland, the question of the export of these materials during the spring melt and autumn freeze-up periods is of interest. Water that left the wetland in early spring was primarily melt water of relatively good quality. Wetland outflows in autumn were relatively small, and treatment was effective up to the time of freeze-up. Water movement was nil during the winter months, except for occasional melt water moving over-ice. Therefore, off-season export of contaminants was minimal.

### 8.5. Early forecasts

For this site and project, there were forecasts made for treatment, based upon field microcosm studies and early project data. The actual performance that took place at this site differed considerably from those pre-project forecasts. For instance, Richardson and Marshall (1986) predicted that this system would become “saturated” after a cumulative loading of  $1.0\text{--}1.5 \text{ gmP/m}^2$ . At present, after 30 years of operation, it is not “saturated” after a cumulative loading of  $56 \text{ gmP/m}^2$

on the 100-ha irrigation area. The reasons for these inaccurate forecasts lie with implicit assumptions that have proven to be incorrect. First, Richardson and Marshall (1986) studied the wrong ecosystem. They evaluated processes for the initial sedge community, which was entirely replaced by a much more productive *Typha* community. Second, they estimated the accretion of organic P in new sediments from short-term  $^{32}\text{P}$  studies and literature estimates, rather than the long-term accretions that reflected the changed community types and degree of fertilization. Third, they assumed that sorption on the initial soils would, in large measure, govern the long-term uptake capacity of the ecosystem. They therefore relied upon laboratory peat core microcosms to characterize the fate of added phosphorus. But in fact, plant uptake and accretion were dominant (Kadlec, 1997). Fourth, they did not realize that wetland hydrology would be greatly altered. They relied upon unconfined embedded mesocosms for plot studies, which in fact were subject to flow through and large advective losses. Further, in the full-scale project, the hydroperiod became 100% for most of the impact zone, and a floating mat developed. Flow is now underneath the *Typha* mat in the majority of the treatment zone. Fifth, they assumed the rate of increase of the impacted area during 1978–1982 would be representative of the long-term behavior of the wetland, but it was representative of only the grow-in period of a few years. Such increases nearly stopped during later years.

These considerations led Richardson (1985) to state: "Collectively, these data indicate that high initial removal rates of phosphorus by freshwater wetlands will be followed by large exports of phosphorus within a few years." However, there was no export data that supported this speculative statement at that time, and subsequent data over the next 25 years, reported here, show no such large exports. To the contrary, the irrigation area removed 94% of the incoming P during the 30-year POR. The growth of the ecological impact zone slowed exponentially (Eq. (11)). The concept of an ever-increasing impact area, evidenced by an ever-moving phosphorus front, did not occur.

#### 8.6. Time trends

The antecedent sorption sites in the wetland apparently became saturated over a period of about 3 years, and stored only about 3% of the added P over the POR (Fig. 11). During the first 9 years, the formation of new biomass had a significant effect on P removal, and stored about 10% of the added P. Thereafter, accretion was the principal mechanism for P removal (Kadlec, 1997, 2009), and stored about 80% of the added P.

These observations have far-reaching implications for understanding and evaluating the potential of various wetland ecosystems for phosphorus removal. Studies of nutrient uptake capabilities of different wetland plants can be very misleading, because short-term growth and storage results are generally not sustainable. In some instances, plant harvest may be contemplated, but that cannot be effective except at very low P loadings (Kadlec and Wallace, 2008). Side-by-side comparisons of plant varieties over a few months are of essentially no value in understanding the long-term

sustainable potential of a wetland containing them. That sustainable potential is governed by accretion, which is the "back end" of the biogeochemical cycle, while uptake is the "front end."

Likewise, sorption studies will characterize that second short-term mechanism of P removal, but it is also not sustainable. Consequently, the determination of sorption isotherms in the laboratory will typically be of little value in characterizing the potential of various wetland ecosystems for phosphorus removal. Sorption results may be used to estimate the length of the initial, enhanced sorption period that may be encountered. Furthermore, the sorbed phosphorus may be desorbed, perhaps in large measure, under some circumstances. For instance, Pant and Reddy (2003) found that sorbed P could be released under laboratory conditions. A small laboratory study showed that the cattail peat at Houghton Lake could desorb about 46% of its total P content into deionized water. This is consistent with measurements of total and HCl extractable phosphorus on soil cores, which averaged 56% extractable in the discharge zone and 44% extractable in control zones. It should be noted that in the wetland environment, any potentially desorbed P is inaccessible to leaching except by diffusion and plant uptake. Porewaters exchange chemicals with overlying waters by diffusion only on the time scale of months, and the process is blocked seasonally by ice formation and the accompanying rejection of solutes. Therefore, the potential release of sorbed P would be mediated by extraction for growth in the root zone, with the nutrients being translocated upward and redeposited on the soil surface with newly forming litter.

Accretion is a long-term sustainable mechanism. Measurements of soil accretion and its P content, although not precise, showed that the removed phosphorus was in new soils and sediments (Kadlec, 2009). Annual accretion was much greater, at  $1.9 \text{ gP/m}^2 \text{ year}$ , occurring at  $5.0 \text{ gP/m}^2 \text{ year}$  during the irrigation season, than that postulated by Richardson (1985), which was a range of  $0.005\text{--}0.24 \text{ gP/m}^2 \text{ year}$ . The Richardson (1985) estimate was based upon a too-low estimate of soil accretion, plus a too-low estimate of soil P concentrations (Kadlec, 2009).

Given a long-term, stable input of phosphorus, the wetland was driven to a condition of long-term, stable performance for P removal. That is the representation afforded by Eqs. (5) and (6). Because the annual calibrations produced uptake rate coefficients with no long-term time trend (Fig. 14), there is evidence that this wetland in no way experienced a "wearing out" phenomenon.

Leakage of small amounts phosphorus to downstream locations in the wetland probably occurred, via the mechanism of episodic floc movement. Sediment movement involves resuspension, advective flow and redeposition. Suspension can be caused by bioturbation, gas release, as well as by shear-induced release of particles from sediments. Settling rates were high (as indicated by lab measurements), and filtration was presumably effective in the dense litter layer. Consequently, this sediment spiraling was of limited magnitude, with annual travel distances estimated to be in the range of 10–100 m. Nonetheless, it may be postulated that this transport mechanism is capable of causing fertilization effects outside the irrigation zone, but on a time scale greater than the 30 years of this study.

### 8.7. Ecological effects

Gross alteration of the original ecosystem occurred in the irrigation zone. The details of this transition are contained in Kadlec and Bevis (2009), and here the general features of the change are only briefly discussed. The responses of the faunal populations to the change are found in Monfils and Prince (2009). Outside the irrigation zone, changes were minimal, and quite likely represented only the slow changes of succession.

As noted earlier, the original vegetation of the wetland in the irrigation area was primarily a sedges, willows, and bog birch. Cattails were present on only seven percent of the test plots in the pre-project wetland. Over the span of the first 9 years of full-scale operation, sedges disappeared from the discharge zone, and were replaced by cattails. The woody shrubs were lost over the ensuing 10 years. The new community of *Typha* contained a number of other plant species, but in small proportions. Over the course of the later half of the 30-year period, the *Typha* community transitioned to a floating mat, and some of the *Typha latifolia* was replaced by *Typha angustifolia*.

The development of the large cattail patch within the larger wetland consequently produced a large fringe zone, or ecotone. The original peatland was habitat for one set of animal populations, and the new cattail patch was favorable habitat for another set, with considerable overlap. Therefore, from the perspective of the entire peatland, this new patch structure fostered higher biodiversity. However, the relative values of the original irrigation zone ecosystem and its replacement may be debated (Kadlec and Bevis, 2009).

## 9. Conclusions

The Houghton Lake community has been well served by the Porter Ranch peatland, which has performed polishing of lagoon-treated wastewater for 30 years. This large wetland has absorbed large quantities of phosphorus, and produced water at background concentrations. The interior zone in which treatment occurs grew throughout the period, and now occupies approximately 100 ha. The growth of this zone has slowed exponentially. A biomachine model provides a good representation of the P removal processes (Kadlec, 1997), which included sorption and biomass expansion in the early years of the project. Throughout the project, the sustainable mechanism of P accretion in new soils and sediments has functioned to immobilize phosphorus. In the later years (10–30), accretion was the only operative mechanism, because sorption and biomass expansion had reached their limits. Over the course of the project, accumulations of 10–30 cm were produced, which contained about 80% of the removed phosphorus. The removal rate model parameter remained stable over the post-startup period, indicating that the wetland displayed no tendency to lose its P sequestration capability. In other words, the wetland showed no signs of “wearing out” or “becoming saturated.”

The irrigation zone, in which the major reductions of contaminants occurred, trapped 53 t of phosphorus over the 30-year POR, for a 94% removal, and did so within a 100-ha area. Because the annual inflow was approximately

700,000 m<sup>3</sup>/year, the hydraulic loading required for this degree of treatment was quite low, averaging an instantaneous seasonal rate of about 0.5 cm/d. Dissolved inorganic nitrogen entered the wetland at about 7.0 mg/L, and was 95% consumed. This amount was insufficient to supply the biomass growth and accretion mechanisms, strongly suggesting that nitrogen fixation provided the majority of the required supply.

The new supply of nutrients to the irrigation zone caused both a fertilizer response and a plant community shift. Biomass in the irrigation zone increased several-fold, and the original sedges were replaced by a cattail-dominant ecosystem. Eventually, this new cattail patch became a floating mat. The community-shift and accretion effects were not anticipated by early prognosticators, whose forecasts of the capacity of the wetland for nutrient removal have turned out to be serious underestimates. The interpretation of long records of treatment wetland performances is a better way to infer what may happen to P-treatment wetlands over the course of time, and such long track records are now being established. The time scale for adaptation at Houghton Lake was 10 years, but subtle ecosystem changes were not complete after 30 years.

## Acknowledgements

The support of the Houghton Lake Sewer Authority was greatly appreciated. Their program of additional research, continued over 30 years, represented a landmark effort to understand and document events in this project. Cooperation and assistance was provided by Brett Yardley, John Hines and Kevin Ackley of the HLSA. Many individuals assisted in field data collection and laboratory analyses, including D.E. Hammer, D.K. Kadlec, R.J. Kruse, H. Alvord, E. Jackson, J. Young.

## REFERENCES

- APHA (American Public Health Association), 1975. Standard Methods of Examination of Water and Wastewater. American Public Health Association, Washington, DC.
- Avanzino, R.J., Kennedy, V.C., 1993. Long-term frozen storage of stream water samples for dissolved orthophosphate, nitrite plus nitrate, and ammonia analysis. *Water Resour. Res.* 29 (10), 3357–3362.
- Bristow, J.M., 1974. Nitrogen fixation in the rhizosphere of freshwater angiosperms. *Can. J. Bot.* 54, 217–221.
- DeGobbiis, D., 1973. On the storage of seawater samples for ammonia determination. *Limnol. Oceanogr.* 18 (1), 146–150.
- Fellman, J.B., D'Amore, D.V., Hood, E., 2008. An evaluation of freezing as a preservation technique for analyzing dissolved organic C, N and P in surface water samples. *Sci. Total Environ.* 392 (2–3), 305–312.
- Gardolinski, P.C.F.C., Hanrahan, G., Achterberg, E.P., Gledhill, M., Tappin, A.D., House, W.A., Worsfold, P.J., 2001. Comparison of sample storage protocols for the determination of nutrients in natural waters. *Water Res.* 35 (15), 3670–3678.
- Haag, R.D., 1979. The Hydrogeology of the Houghton (Lake) Wetland. MS Thesis, University of Michigan.
- Hammer, D.E., Kadlec, R.H., 1986. A model for wetland surface water dynamics. *Water Resour. Res.* 22 (13), 1951–1958.
- Hammer, D.E., 1984. An Engineering Model of Wetland/Wastewater Interactions. PhD Thesis, University of Michigan.

- Hanna, E., McMullan, B., McGrenere, P., 2001. A full-scale, full-treatment constructed wetland for cobalt, Ontario. In: Proceedings, 29th Annual Water Environment Association of Ontario Technical Symposium, Toronto, Ontario.
- Headley, T., Kadlec, R.H., 2007. Conducting hydraulic tracer studies in constructed wetlands: a practical guide. *Ecohydrol. Hydrobiol.* 8 (3/4), 269–282.
- Herskowitz, J., 1986. Listowel Artificial Marsh Project Report. Ontario Ministry of the Environment, Water Resources Branch, Toronto, 251 pp.
- Kadlec, R.H., Li, X.-M., 1990. Peatland ice/water quality. *Wetlands* 10 (1), 93–106.
- Kadlec, R.H., Knight, R.L., 1996. Treatment Wetlands. CRC Lewis Publishers, Boca Raton, FL.
- Kadlec, R.H., Wallace, S.D., 2008. Treatment Wetlands, 2nd ed. CRC Lewis Publishers, Boca Raton, FL.
- Kadlec, R.H., 1984. Freezing-induced vertical solute movement in peat. In: Seventh International Peat Congress Proceedings, Dublin, Ireland, pp. 248–262.
- Kadlec, R.H., 1990. Overland flow in wetlands: vegetation resistance. *ASCE Journal of Hydraulic Engineering* 116 (5), 691–706.
- Kadlec, R.H., 1993. Natural wetland treatment at Houghton Lake: the first fifteen years. In: Proceedings of the WEF 66th Annual Conference, Anaheim, CA, WEF, Alexandria, VA, pp. 73–84.
- Kadlec, R.H., 1997. An autotrophic wetland phosphorus model. *Ecol. Eng.* 8 (2), 145–172.
- Kadlec, R.H., 1999. Chemical, physical and biological cycles in treatment wetlands. *Water Sci. Technol.* 40 (3), 37–44.
- Kadlec, R.H., 2003. Effects of pollutant speciation in treatment wetland design. *Ecol. Eng.* 20 (1), 1–16.
- Kadlec, R.H., 2008. The lagoon-wetland treatment system at Onaway, Michigan. In: Proceedings, 37th Annual Water Environment Association of Ontario Technical Symposium, Collingwood, Ontario.
- Kadlec, R.H., 2009. Wastewater treatment at the Houghton Lake wetland: soils and sediments. *Ecol. Eng.* 35, 1333–1348.
- Kadlec, R.H., Bevis, F.B., 2009. Wastewater treatment at the Houghton Lake wetland: vegetation response. *Ecol. Eng.* 35, 1312–1332.
- Kadlec, R.H., Hammer, D.E., 1988. Modeling nutrient behavior in wetlands. *Ecol. Model.* 40, 37–66.
- Kadlec, R.H., Alvord Jr., H.H., 1989. Mechanisms of water quality improvement in wetland treatment systems. In: Fisk, D.W. (Ed.), *Wetlands Concerns and Successes*. AWRA, pp. 489–498.
- Kadlec, R.H., Hammer, D.E., Nam, I.-S., Wilkes, J.O., 1981. The hydrology of overland flow in wetlands. *Chem. Eng. Commun.* 9, 331–344.
- Kadlec, R.H., Williams, R.B., Scheffe, R.D., 1987a. Wetland evapotranspiration in temperate and arid climates. In: Hook, D.D. (Ed.), *The Ecology and Management of Wetlands*. Croom Helm, Beckenham, pp. 146–160 (Chapter 12).
- Kadlec, R.H., Li, X.-M., Cotton, G.B., 1987b. Modeling solute segregation during freezing of peatland waters. *Water Resour. Res.* 24 (2), 219–224.
- Klingaman, E.D., Nelson, D.W., 1976. Evaluation of methods for preserving the levels of soluble inorganic phosphorus and nitrogen in unfiltered water samples. *J. Environ. Qual.* 9 (1), 42–46.
- Li, X.-L., 1985. Solute Rejection During Freezing of a Peatland and Overlying Water. PhD Thesis, University of Michigan.
- MacDonald, R.W., McLaughlin, F.A., 1982. The effect of storage by freezing on dissolved inorganic phosphate, nitrate and reactive silicate for samples from coastal and estuarine waters. *Water Res.* 16, 95–104.
- Maguire, L.A., 1974. A Model of Beaver Population and Feeding Dynamics in a Peatland at Houghton Lake, Michigan. MS Thesis. The University of Michigan, Ann Arbor, 127 pp.
- Marvin, K.T., Proctor Jr., R.R., 1965. Stabilizing the ammonia-nitrogen content of estuarine and coastal waters by freezing. *Limnol. Oceanogr.* 10 (2), 288–290.
- McDaniels, A., 1979. Virus Survey of Sewage Plant Holding Ponds and Swamp Area of Effluent Discharge of Houghton Lake, Michigan. Report of NSF-RANN Project, School of Public Health, University of Michigan, Ann Arbor, MI.
- Monfils, M.J., Prince, H.H., 2009. Breeding bird use of a Northern Michigan peatland utilized for wastewater Treatment. *Ecol. Eng.* 35, 1357–1365.
- Nearhoof, F.L., 1996. A protocol for collecting surface water samples in marshes of the Florida everglades. *Water Quality Technical Series*, vol. 3, no. 25. Florida Department of Environmental Protection.
- Pant, H.K., Reddy, K.R., 2003. Potential internal loading of phosphorus in a wetland constructed in agricultural land. *Water Res.* 37 (5), 965–972.
- Pecor, C.H., Novy, J.R., Childs, K.E., Powers, R.A., 1973. Houghton Lake annual nitrogen and phosphorus budgets. Technical Bulletin No. 73-6, Department of Natural Resources, Lansing, MI.
- Phillips, J., Pries, J., 2001. Town of Brighton: a treatment wetland success story. In: Proceedings, 29th Annual Water Environment Association of Ontario Technical Symposium, Toronto, Ontario.
- Price, D.S., Smith, D.W., Stanley, S.J., 1995. Performance of lagoons experiencing seasonal ice cover. *Water Environ. Res.* 67 (3), 318–326.
- Richardson, C.J., Marshall, P.E., 1986. Processes controlling movement, storage, and export of phosphorus in a fen peatland. *Ecol. Monogr.* 56 (4), 279–302.
- Richardson, C.J., 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. *Science* 228, 1424–1427.
- Richardson, C.J., Kadlec, J.A., Wentz, W.A., Chamie, J.P.M., Kadlec, R.H., 1976. Background ecology and the effects of nutrient additions on a central Michigan wetland. In: LeFor, M.W., Kennard, W.C., Helfgott, T.B. (Eds.), *The Proceeding of the Third Wetland Conference*. Institute of Water Resources, The University of Connecticut, pp. 34–72, Report No. 26. Also NTIS PB2543367.
- Richardson, C.J., Tilton, D.L., Kadlec, J.A., Chamie, J.P.M., Wentz, W.A., 1978. Nutrient dynamics of northern wetland ecosystems. In: Good, R.E., Whigham, D.F., Simpson, R.L. (Eds.), *Freshwater Wetlands: Ecological Processes and Management Potential*. Academic Press, New York, pp. 217–242.
- Scheffe, R.D., 1978. Estimation and Prediction of Summer Evapotranspiration from a Northern Michigan Wetland. MS Thesis, University of Michigan.
- Schwiegler, B.R., 1978. Effects of Sewage Effluent on Algal Dynamics of a Northern Michigan Wetland. MS Thesis. The University of Michigan, Ann Arbor, 53 pp.
- Tilton, D.L., Kadlec, R.H., 1979. The utilization of fresh-water wetland for nutrient removal from secondarily treated waste water effluent. *J. Environ. Qual.* 8 (3), 328–334.
- Waughman, G.J., Bellamy, D.J., 1980. Nitrogen fixation and the nitrogen balance in peatland ecosystems. *Ecology* 6 (5), 1185–1198.
- Williams and Works, 1976. Houghton Lake Area Step I Facility Plan. Report to The State of Michigan in support of Application for a Construction Grant from USEPA, April 1976.