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Wastewater treatment at the Houghton lake wetland: Soils and sediments

Robert H. Kadlec

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

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ABSTRACT

This paper describes the sediment and soils responses in a very long-running study of the capacity of a natural peatland to remove nutrients from treated wastewater. Data are here presented and analyzed from three decades of full-scale operation (1978–2007), during which large changes in the wetland soils occurred. An average of 600,000 $m^3 y^{-1}$ of treated water was discharged each warm season 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. Nutrients were stored in the 100 ha irrigation area, which removed 94% of the phosphorus (53 t) and 95% of the dissolved inorganic nitrogen. Phosphorus was stored in new biomass, increased soil sorption, and accretion of new soils and sediments, with accretion being dominant. Peat probings, water level increases and topographical surveys established quantitative measures of soil accretion. Over 30 cm of new soil developed, in which nutrient storage occurred. Phosphorus concentrations in the new soil were approximately 2000 mg P/kg, and the nitrogen concentration was 2–3%DW. The removal of TSS was effective, but minor in comparison to the internal generation and cycling of produced particulates. Later in the project history, the interior portion of impacted area became a floating mat. Sedimentation processes then occurred with no exposure to above-mat detrital processes. Trace element analyses showed no appreciable accumulation of heavy metals, other than the calcium and iron that characterized the antecedent wetland and the incoming water. Biomass cycling models were found to produce reasonable estimates of the measured nutrient accumulations. The light loadings of nutrients to this system produced dramatic effects in the ecosystem, but were lower than the range seen in some other treatment wetlands. Insufficient nitrogen was added to support the new biomass, and nitrogen fixation was identified as a possible compensatory mechanism.

1. Introduction

Details of the project history and other aspects of the project have been described in Kadlec (2009a,b). Here only a brief summary of the principal features of the project is given. Wastewater from the Houghton Lake community is treated in two aerated lagoons, and stored in a third pond during the cold half of the year. This treated water is transferred during the summer half-year to a smaller pond, and thence to an existing peatland located about 2 km from the ponds.

The Porter Ranch wetland comprises approximately 700 ha in its entirety. The zone of interest for water quality improvement and vegetative impacts was a smaller zone near the discharge. That zone contained leatherleaf-bog birch and sedge-willow cover types. Standing water was usually present in spring and autumn, but the wetland had no surface water during dry summers. 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 meters thick. Interior flow in the wetland occurs by overland flow, proceeding from 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 surface sheet flow.

The wastewater was fully treated inside an irrigation zone of approximately 100 ha surrounding the discharge pipeline (Fig. 1). The hydraulic loading was approximately 600,000 $m^3 y^{-1}$, applied during the pumping season, which averaged 128 days in length, during May–October. The effluent contained 3.5 mg/L of total phosphorus (TP), and 7 mg/L of dissolved inorganic nitrogen ($DIN = NH_4N + NO_XN$). The annualized rate of TP addition was 1.87 $g P m^{-2} y^{-1}$, and 0.11 $g P m^{-2} y^{-1}$ was exported from the irrigation zone. The irrigation area removed a total of 53 metric tons (t) of phosphorus over the 30-year period of record (POR). The aver-

E-mail address: rhkadlec@chartermi.net.

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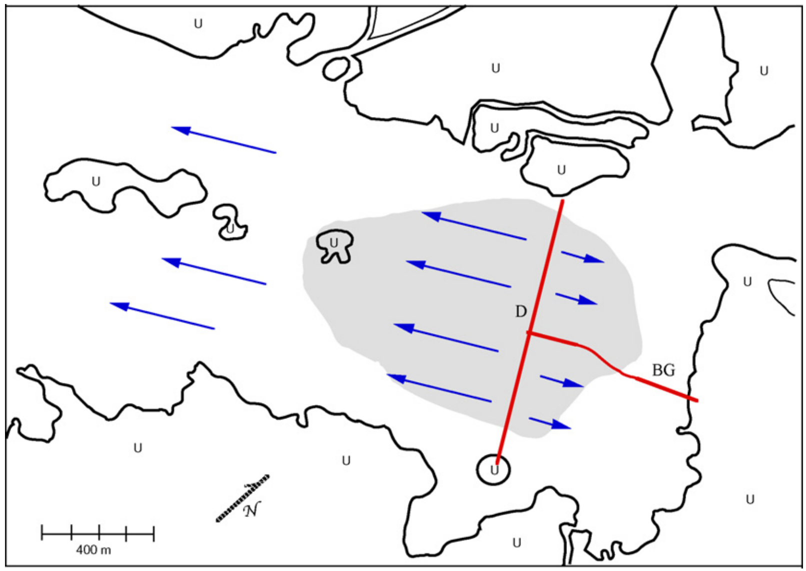

Fig. 1. Outline map of the northeastern end of the Porter Ranch peatland. The discharge line extends from southeast to northwest, and receives water from the transfer line from the east. U = Surrounding upland. The shaded area represents the approximate zone of impacts. Discharge locations are labeled D, and backgradient locations are labeled BG.

age annual loading of dissolved inorganic nitrogen to the irrigation zone was 4.50 g N m^{-2} y^{-1}, and only 0.11 g N m^{-2} y^{-1} left the irrigation zone. A total of 131 t of dissolved inorganic nitrogen (DIN) were removed in the wetland irrigation area over the 30-year period. Details of this and other water quality improvement may be found in Kadlec (2009a). In addition to the irrigation zone, the terminology used here identifies a discharge zone, which was in close proximity to the discharge pipeline (i.e., within 100–200 m). As a reference, control zones were examined, which were remote from the discharge zone.

The water addition was about double the amount of rainfall experienced by the irrigation zone. More importantly, the added water on average more than compensated for evapotranspiration. The original peatland was dry during most summers, but had surface water during autumn, winter and spring. As a result, the hydroperiod of the irrigation area was extended, from about 50% to 100% in the warm 6 months of the year. There were consequently two different impact zones: one that experienced high nutrients and extended hydroperiod, near the discharge; and a second that experienced the long hydroperiod, but not excess nutrients, just outside the region in which nutrient stripping occurred. These are termed the discharge zone and the backgradient zone respectively.

The soils within the irrigation zone were affected by the extra water and nutrients, but the remainder of the peatland did not experience changes. It is the purpose of this paper to document the changes in the Porter Ranch peatland soils and sediments that occurred as a consequence of the addition of the treated wastewater.

2. Experimental methods

A number of procedures were utilized to assess soils and soil-building processes during the project history. Most of these were utilized only in a subset of the total of 30 years of operation. The pre-project methodologies have been described elsewhere (Chamie, 1976; Wentz, 1975; Richardson et al., 1976). New methods were developed as needed for the marsh environment.

Sampling locations are identified on a reference grid, indicating the distance from the discharge pipeline in meters and a letter designation for the transect line parallel to flow. Letters are from “A” 400 m north of the tee, to “C” passing through the tee, to “E” 400 m south of the tee (Fig. 1). Most samples for soils analysis were taken on the “C” line, at various distances from the tee.

2.1. Suspended solids

There is no standard or commonly accepted method for sampling suspended solids in shallow-water wetland environments. The combination of shallow-water and easily disturbed sediments causes unacceptable performance of “standard” samplers and sampling procedures. For instance, the removal of a 1 L sample from a water sheet 10 cm deep over 1-min period causes local velocities which are an order of magnitude greater than the average water sheet velocity. This causes resuspension of settled material, leading to errors of one or more orders of magnitude. The boots of the sample-taker also create displacements and resuspension.

These problems led to the development of a passive sampler that 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 (see Fig. 2A). The valves were constructed to close with minimum force in one quarter turn of the wire handle. A 1-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 endpience 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 in the proper location, using the empty tube as the positioning figure. The frozen 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 vacated so that the disturbed sediments could resettle and be flushed downstream. Since settling times and water velocities exceed the melting time, the tube did not open to flushing until the disturbance 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

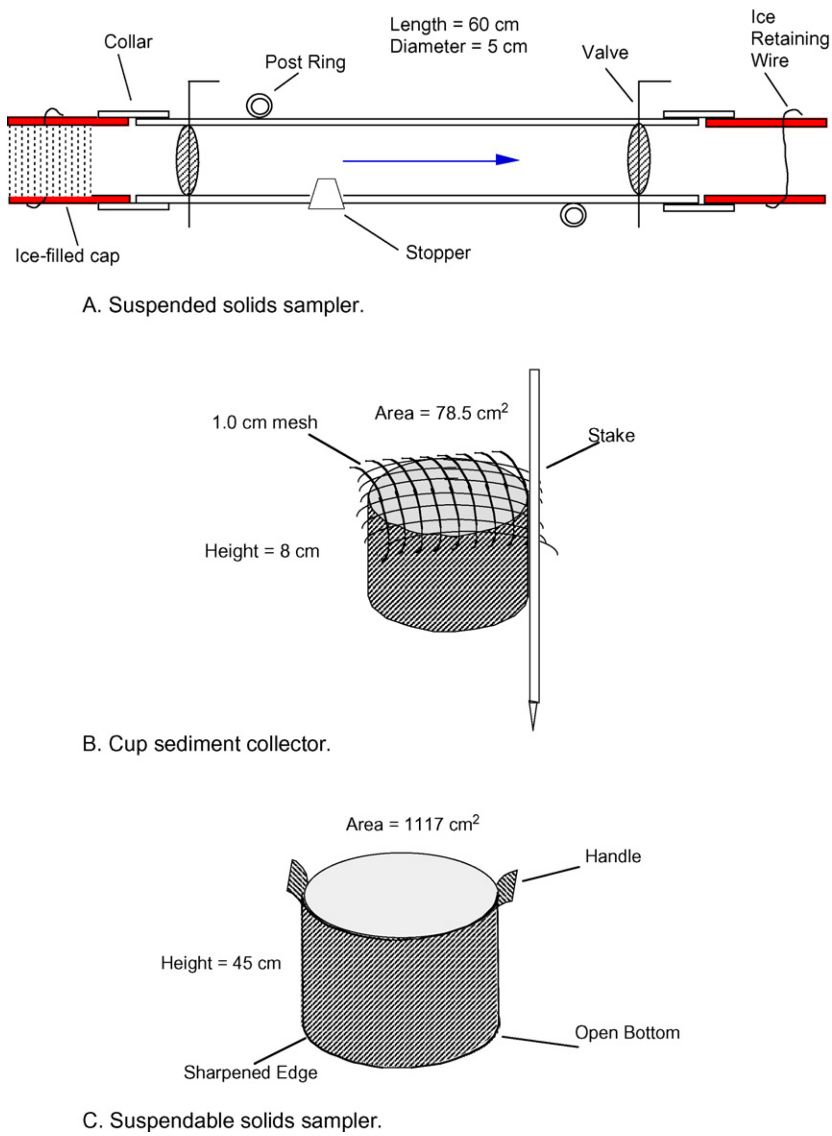

Fig. 2. (A) Suspended solids sampler, (B) cup sediment collector and (C) suspendable solids sampler.

and the contents transferred to a sample bottle via the unstoppered side hole. The samples were subsequently filtered and dried in the laboratory, following standard methods (APHA, 1975).

2.2. *Sedimentation flux*

The vertical flux of solids reaching the soil surface was measured using cylindrical sediment cups of 0.0785 m^2 area and 8 cm in depth (see Fig. 2B). Since it was impossible to place these in the bottom sediments without severe and random sediment disturbance, the cups were initially filled to the brim with tap water and frozen. The resultant domed ice plug prevented accumulation of the sediments stirred up by placement. The dome shape, plus the melting of the top, flushed the ice cap. One centimeter mesh plastic netting was taped over the top of each cup before freezing. This served two purposes: first, to prevent the ice plug from releasing from the cup; and second, to screen out large objects. The cups were stapled to stakes driven into the peat soil, with the cup rim at least 3 cm above the sediment surface. This gave sufficient freeboard to contain the downward solids flux without resuspension. Harvest was accomplished by approaching the cup carefully from the downstream side and pulling the stake. The cups were not selectively colonized by any macroinvertebrate, but beavers did display a preference for the stakes (and cups) as structural material.

Sediment collectors of this design have been successfully used in marsh environments (Jordan and Valiela, 1983). They have been found to be accurate sedimentation integrators at low velocities (<10 cm s^{-1}) (Gardner, 1977, 1980; as reported by Jordan and Valiela, 1983). Samples were returned to the laboratory, dried and weighed (APHA, 1975).

Sample locations were at 100 m intervals along downgradient transects in each of the three cover types: sedge (*Carex* spp.), leatherleaf (*Chamaedaphne calyculata*) and cattail (*Typha* spp.). Samples were taken at monthly intervals during the unfrozen season, during 1981.

2.3. *Suspendable bed solids*

Replicate plots were isolated and harvested for suspendable material. A stainless steel cylinder, with a sharpened bottom edge and with handles attached to the top edge, was twisted down into the peat, thus isolating 0.111 m^2 of surface water, vegetation and suspendable material (see Fig. 2C). Duckweed (*Lemna* spp.) was gently skimmed from the interior using a wire mesh strainer. Emergent vegetation was clipped and discarded along with plant litter so that the sediments could be stirred. The water depth was measured at least three locations. The water was stirred gently, to suspend solids, including phytoplankton, but not violently enough to detach periphyton. A water sample was then immediately taken. A subsample was returned to the laboratory, filtered, oven dried at 80 °C and weighed. A second subsample was centrifuged, filtered and extracted with acetone for chlorophyll determination, according to standard methods (APHA, 1975).

2.4. Soil cores

Soils were sampled by coring, using several methods. Early in the project history, it was possible to push and twist a sharpened plastic tube, cutting through roots and soils. The tube was then stoppered to provide air-lock suction, and pulled vertically upward. The soils were then extruded from the plastic tube, and sectioned into desired lengths to establish vertical profiles. Prior to the start of irrigation, soils were well-consolidated, and cores suffered little of no compression inside the pushed tube. However, during the 1980s, it became apparent that the soils in some areas had become less well-consolidated, and core compression became more important. For that period, a compression correction was applied to relate the section positions to the original soil horizons. However, by 1985 there was insufficient soil strength to extract push tube cores, which flowed out of the tube upon extraction.

A substitute technique was developed, which worked quite successfully. An aluminum pipe (1.22 m × 3.8 cm × 1.6 mm wall), with a sealed solid tapered point (13-cm long), was gently pushed into the wetland. It was then filled with a mixture of dry ice and acetone (−80 °C), which caused exterior freezing of peat, litter, water, or any proportions thereof. A freezing time of 15–20 min typically produced a frozen plug of outer diameter 12–16 cm. The pipe and frozen annulus were removed from the wetland, and the annulus removed by filling the tube with hot water. This caused melting of a thin layer of frozen soil adjacent to the pipe, and allowed the pipe to be removed from the frozen soil plug. The resulting annular core was then sectioned as desired with a saw.

Bulk density was calculated from the dimensions of soil sections and the corresponding dry weight, for cores taken in 2006. It was calculated from estimated sample volume and dry weight for other cores.

A floating mat developed in the central portion of the irrigation zone. Core sampling of the mat was done by cutting through the roots and soils with a machete, freeing plugs of material. Plugs of 20 cm × 20 cm, and length 25–40 cm were then lifted vertically upward. These were then sectioned with a machete. These plugs unavoidably partially drained after extraction, precluding determination of bulk density. This technique was also successfully applied in control zones outside the floating mat zone, in which the cut plugs could be lifted free of the subsoils beneath the root zone.

2.5. Topographical surveys

Topographical surveys were conducted on several occasions, to determine the elevation of the wetland surface. The first of these was done in 1972, 6 years prior to any wastewater additions. That survey included the entire 700 ha peatland. A 400 m grid was used, with points at the grid intersections and approximately four associated tuning point locations per grid. The datum was the top of a concrete dam structure at the wetland outlet. This survey was done in winter, and the vertical surface was the top of the frozen peat. In a few instances, it was the ice surface on pooled water. During this survey, several solid steel stakes were driven through the peat and into the mineral soils below, to serve as future reference elevations. A second survey was done in 1977, as part of the engineering of the full-scale wastewater distribution system. The surveyed area was about 200 ha, including the eventual 100 ha irrigation zone. This was also done in winter, at which time the vertical surface was the top of the frozen peat. The datum was a benchmark at a nearby road intersection. A third survey was done in 1979, at which time a 200 m grid of nineteen stations was set in the irrigation area. A total of 41 soil and water elevations were determined (see Fig. 6). This survey was done in March, at which time there was 25 ± 9 cm of standing water. The datum was the top of a steel stake in the irrigation area, set during the 1972 survey. Very little water had been discharged in 1978, and therefore the 1979 survey was deemed to represent pre-project conditions.

A small region of the irrigation area, along a centerline from 400 m backgradient of the tee, to 100 m downstream of the tee, was resurveyed in 1988, 10 years after irrigation commenced. The datum was again the top of a steel stake in the irrigation area, set during the 1972 survey. Surveys were done in winter, with ice present, and again in early spring, after ice melt. In winter 2005, the irrigation area was resurveyed using modern satellite techniques, with five replicates at each location. Twenty stations were located precisely, and marked with steel pipes driven into mineral soil (see Fig. 7). The Deadhorse Dam elevation was also used as a reference in this survey.

2.6. Chemical composition

Ash content represents the mineral fraction of the soil, as determined by removal of the organic matter by burning at 550 °C. The burned fraction is termed the volatile fraction. Some chemicals, notably metals, are retained predominantly in the ash fraction. In addition to sectioned soil cores, samples of suspendable sediment were also ashed. Grab samples and cup samples were dried and ground to produce uniform powders. A subsample was digested using H_2SO_4 and H_2O_2 (Van Lierop, 1976). Phosphorus in the digests were determined by the ascorbic acid–molybdate procedure (APHA, 1975), and nitrogen was determined using an ammonium specific ion electrode (Orion, 1983). The Soil Science laboratories of the University of Florida also performed nitrogen and phosphorus determinations. A second subsample was analyzed for sulfur by combustion and acid–base titration using an analyzer by LECO™. A third set of subsamples was analyzed for calcium, magnesium and copper using atomic adsorption spectrometry (Perkin-Elmer, 1982). A fourth subsample was analyzed for 31 other elements using neutron activation analysis.

3. Calculations

The mass balances associated with plant growth have been documented in Kadlec (2009a,b) in this issue, and will not be repeated here. Additionally, mass balance calculations were performed for the sedimentation experiments and for sorption, as detailed here.

3.1. Surface solids mass balances

Definition of the generation, movement and location of solids is facilitated by interpretation via material balances. It is difficult to “model” the contributing processes in detail, but even gross approximations can yield informative results. Of prime importance is the use of the mass balance to establish the overall, annual suspended solids budget. The simple calculation procedure shown below is site specific, and explains no internal details of the component processes.

The amount of sediment, the concentration of suspended solids, and the amount of cup-collected sediment are functions of time, distance. They are related by transient balances for solids in the water and in the cup collectors. It is desirable to interpret the data to produce estimates of the rates of the fundamental processes of sedimentation (S), resuspension (R), net generation (G), and lateral transport.

A balance equation may be written for solids for in the water. The water column equation is:

$$h\frac{\partial C}{\partial t} + uh\frac{\partial C}{\partial x} = G + R - S \tag{1}$$

where C = concentration, g m^{-3} = mg/L; G = generation rate, g m^{-2} d^{-1}; h = water depth, m; R = resuspension rate, g m^{-2} d^{-1}; S = settling rate, g m^{-2} d^{-1}; t = time, d; u = superficial water velocity, m d^{-1}; x = distance, m.

The changes in inventory in the water column may be assumed to be small compared to the lateral flux, generation and settling. Therefore, a quasi-steady mass balance will suffice for estimating purposes:

$$uh\frac{\partial C}{\partial x} = G + R - S \tag{2}$$

where a one-dimensional flow has been assumed.

The overall sinking flux may be represented as the product of the settling velocity and concentration for dilute, non-interacting suspensions:

$$S = k_S C \tag{3}$$

The detachment and resuspension of solids is a similarly complicated process. Several procedures are available in the soil erosion literature, including the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1965) and its modifications. Detachment due to overland flow is generally believed to be related to velocity and to the soil erosivity (Ross et al., 1980; Yalin and Kayahan, 1979). Laboratory experiments with the wetland sediments verified the velocity behavior for the Houghton Lake sediments. A nearly constant velocity situation prevailed, because of the constant pumping rate. The resuspension rate was therefore taken to be an empirical zero-order rule; R = constant.

The generation of sedimentary material was found to be the most important rate term in the irrigation area. The generous supply of nutrients in this zone assures a large production of a wide variety of transportable organisms and associated dead organic material. The irrigation area was characterized by high water chlorophyll content, and high sediment accumulation. In the absence of detailed data on production and decomposition of this material, a constant two-level net generation rate was used:

$$G = \left\{ \begin{array}{ll} G_{\max}, & \text{if nutrients} \\ G_{\min}, & \text{if no nutrients} \end{array} \right\} \tag{4}$$

The nutrient-rich zone extended from the pipeline (x = 0), downstream to a distance x = L, at which point the phosphorus and nitrogen were essentially gone from the water. Because there was no independent means of determination of resuspension and generation, only the combination, R + G, was used in calibration.

The amount of solids collected in a cup trap is presumed to be due to settling only. Therefore, the cup inventory is the result of accumulating the settling flux over the duration of collection at any given point along the direction of travel:

$$\frac{\partial M}{\partial t} = S = k_S C \tag{5}$$

where M = mass collected in cup, g m^{-2}.

The amount of solids collected in a cup trap is presumed to be due to settling only. Therefore R + G = 0, and Eq. (2), applied to the cup traps, becomes:

$$uh\frac{\partial C}{\partial x} = -k_S C \tag{6}$$

3.2. *Sorption of nutrients*

Peat soils have sorptive capacity for phosphorus, although this storage is soon saturated under any increase in phosphorus loading (Kadlec and Wallace, 2008). A number of different sorption isotherms have been proposed, with the Freundlich formulation among them:

$$S = KC^n \tag{7}$$

where C = P concentration in water, mg/L; K = proportionality constant (mg P/kg)/(mg/L)^n; n = fitted exponent; S = sorbed P concentration, mg P/kg.

This isotherm was used to calculate the amount of sorbed phosphorus in the P mass balance.

4. Soil and sediment amounts

No detailed pre-project studies were conducted relative to the amounts of sediments or soils present in the peatland, because there was not an appreciation of the importance of accretion in nutrient removal. The peat thickness was typically 1–3 m. The bottom of the deep peat was carbon dated at 773 ± 130 years before present, for a life-average accretion rate of approximately 0.25 cm y^{-1}. The radio cesium peak (deposited ca. 1950) was found about 4 cm below the surface in the sedge cover type in 1980, indicating a modern, pre-project accretion rate of about 0.12 cm y^{-1}. Field observations indicated that wetland sediments increased due to irrigation, and that significant peat accretion occurred over the 30 years.

The peatland is located in a shallow depression in the glacial drift of the region, which has a thickness of 150–200 m and is situated on bedrock (Coldwater shale). Immediately under the peat layer is a thick layer of impervious blue clay. Soil borings were conducted as part of the pre-project investigations, to depths of 15–50 m. Only 3 of 12 borings passed through the clay, and these showed a mean thickness of 3 m. The remaining nine borings did not penetrate the clay entirely, but were on average 3 m into clay at termination. Haag (1979) studied the 100 ha irrigation area, using electrical resistivity methods to establish layer thicknesses. Fifty stations in the irrigation area showed a mean clay thickness of 4 m. Haag (1979) concluded that the clay layer was continuous under the wetland, and conducted lysimeter studies that demonstrated that water did not move vertically through this clay layer.

The peat layer that formed the top-most soil horizon was also continuous across the entire wetland. This material was about a meter thick, and could easily be penetrated by hand-driven pole probes. Consequently, several pre-project peat thickness surveys were conducted. A 1972 marshwide probing (700 ha) showed 1.34 ± 0.94 m (mean ± S.D.) at 27 stations. In the 100 ha irrigation area, Haag (1979) found 0.81 ± 0.44 m by probing, and 0.79 ± 0.47 m by electrical resistivity, both at 50 stations. The design of the pipeline support dock required knowledge of the peat depth, so as to specify the required lengths of support stakes that held the dock and pipeline. Those support stakes were specified to be driven down into the clay layer for stability. A peat probe survey was therefore conducted in 1977, following the entire route of the dock, both the leader portion out to the tee, and the discharge section that held the gated irrigation pipe (Fig. 1). This route survey showed 0.84 ± 0.44 m of peat for 180 stations. These pre-project peat thicknesses served to define the antecedent conditions in the wetland. These soil conditions were altered by the irrigation, and studies were subsequently conducted to gain some understanding of mechanisms.

4.1. Suspended solids

Project permits required monitoring of total suspended solids (TSS) in the lagoon discharge, and for the last several years of the 30-year period, a limit of 30 mg/L was imposed on the lagoon discharges. The overall period of record TSS from the lagoon was 34 mg/L. However, for 1993 and prior, the average was 17 ± 4 mg/L (mean ± S.D.), and for 1994 and subsequent, the average was 50 ± 20 mg/L. As a consequence, there were numerous exceedences of the permit monthly limit of 30 mg/L during the last 14 years. Over the 30-year period of record (POR), 633 t of TSS were added to the wetland via the pumped treated wastewater. For the 100 ha irrigation area, the average loading was 0.15 g DW m^{-2} d^{-1} for the entire POR. At a bulk density of 0.2 g cm^{-3}, this would amount to only 3.2 mm of accretion over the 100 ha irrigation area. However, the wetland could have caused setting and filtration in a small fraction of that area. If removal occurred in a 10 ha zone, then the POR accretion would have been 3.2 cm. These deposition rates are so low that it is highly likely that plants would have no difficulties due to sediment smothering, and the roots would have been able to easily keep pace with the rate of accretion of incoming sediments.

It was recognized from the start that TSS was very difficult to sample at internal wetland stations, because of the shallow-water (ca. 15 cm) and the loose, flocculent character of bottom sediments. Consequently, TSS was not routinely monitored at interior locations. 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 ± 3 mg/L, while within the zone near the discharge, TSS = 34 ± 28 mg/L. During that season, the pumped TSS = 22 ± 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 water 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. This planktonic algae was not present at downstream locations, as confirmed by chlorophyll measurements on water samples (Kadlec, 2009a). 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.

4.2. Suspendable solids

The layer of particulate material, situated in and on the pronate litter, was considerably greater than the suspended material above it in the water column. The accumulation of several years sedimentation, less decomposition, form this unconsolidated layer, which may be termed "floc." Measurements were made of the amounts of suspendable material on transects parallel to flow in the sedge cover type in 1980 and 1981. There was no clear spatial pattern, with the average for the irrigation area of 718 ± 389 g DW m^{-2}, and for control areas, 1087 ± 399 g DW m^{-2}. To place these amounts in perspective, the measured solids concentrations in the gently stirred water were 5500 ± 2700 mg/L for the irrigation area, and 8700 ± 1100 mg/L for control areas. The amounts suspended by this stirring technique were not expected to be the same as those that are susceptible to entrainment by natural processes. However, it was an indicator of the relative amounts of suspendable material.

A second measure of the amount of floc was the solids found in the water column for soil cores taken by the freezing technique. The overlying water was also frozen in this process, and analyzed for solids content. For six cores in the irrigation area, the water layer contained 1.8 ± 1.0%DW solids, or approximately 1800 mg/L. For seven cores in control areas, the water layer contained 1.7 ± 1.2%DW solids, or approximately 1700 mg/L.

Both measurement techniques found very large amounts of mobile material at both irrigation area and control locations. By implication, sediment movement by settling and resuspension were important mechanisms for solids redistribution in the wetland.

4.3. Settling solids

The vertical settling of particulates was reflected in the amounts of material accumulated in the cup collectors. The results for three vegetation cover types are shown in Table 1. The results are gross settling rates, because resuspension was largely suppressed by the relatively deep side walls of the cups. At the time of these samplings, the cattail monoculture had not expanded to exclude other vegetation types in the irrigation zone. The most relevant sedimentation rates are those for the cattail community, which eventually dominated the irrigation area.

Data was for the mean amount of material found in duplicate cups at various locations at various collection times. The time trends at any particular location were roughly linear, and therefore linear regression was used to establish the mean deposition rate over the entire irrigation season. The R^2 values for the various locations were in the range 0.72–0.92 (Table 1). Cattail zones had deposition rates of 3.08 g DW m^{-2} d^{-1} in the irrigation area, and 0.62 g DW m^{-2} d^{-1} in control areas. The sedge-willow cover type had slightly lower deposition rates, 2.11 and 0.51 g DW m^{-2} d^{-1} d in irrigation and control zones respectively. As a reference, the TSS loading to the irrigation area in 1981 was 0.071 g DW m^{-2} d^{-1}. The

Table 1
Sediment collection rate in cup collectors at various locations in the wetland. Duplicate cups were harvested on four dates during the 1981 summer irrigation season.

Distance (m)	Sedge-willow ($g\,m^{-2}\,d^{-1}$)	Cattail ($g\,m^{-2}\,d^{-1}$)	Leatherleaf–bog birch ($g\,m^{-2}\,d^{-1}$)
Irrigation area			
0	1.66	1.38	1.22
80		5.69	
100			1.11
150	1.73		
160		2.16	
200			0.76
220		3.09	
250	3.60		
275			0.80
350	1.44		
Irrigation mean	2.11	3.08	0.97
Control mean	0.51	0.62	0.31
Average R^2	0.72	0.85	0.92

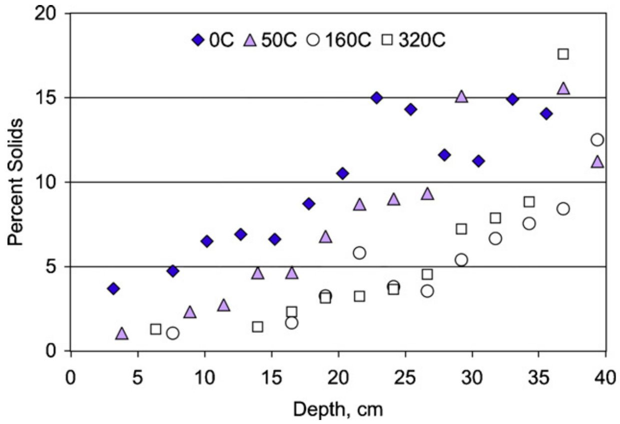

Fig. 3. Example solids density profiles as a function of depth below the water surface in 1987. Depth is below water surface. Points are identified by distance from discharge (m) and the central transect line (*C*).

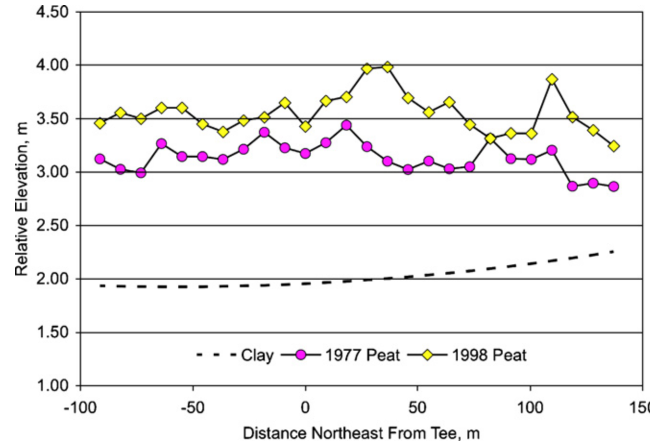

Fig. 4. Increase in soil elevation over the first 20 years. Distance is along the discharge pipeline, from 100 m southwest to 150 m northeast.

amount of gross sedimentation exceeded the incoming TSS loading by a factor of 43 in the cattail area.

4.4. Soil cores

In several years, peat cores were obtained, backgradient from the discharge line. Solids density–depth profiles at each location were obtained. Sample results from 1987 are shown in Fig. 3, from which it appears that there is about a 12–14 cm difference in the elevation of a particular solid percentage between the discharge area (0C and 50C) and backgradient area (160C and 320C). The reference for these depths was the top of the water. This shift in the profiles may be further understood if the depths are measured from the top of the soil, although that horizon was poorly defined. There was essentially no difference in the solids content, either as bulk density or percent solids, of the top two 10-cm layers of soil in discharge and control zones (Table 2). However, the layer at 20–30 cm soil depth shows low solids content in the discharge zone, compared to the control zones. This was likely due to the process of floating mat development, in which the root mat separated from the soils below. The region of low solids at 20–30 cm may have been located below the root mat, which floated to a higher elevation.

In 2006, the solids density profiles were much more uniform (Table 2). In the discharge area, the core was entirely in the floating mat, which had a fairly uniform solids profile. The soil material was black in color, and was absent below the root zone. The mat thicknesses were 29.8 ± 1.7 cm.

Table 2
Solids bulk density and percent solids in discharge and control zones, 1987–1989 and 2006 (g/cm^3). The designation "control" means outside the zone of nutrient impact.

Location	Percent solids			Bulk density	
	Depth	Mean	S.D.	Mean	S.D.
1987–1989					
Discharge	0–10	3.72	1.61	0.038	0.016
	10–20	6.88	3.41	0.070	0.035
	20–30	17.37	11.37	0.181	0.125
Control	0–10	3.45	1.39	0.035	0.014
	10–20	6.68	2.08	0.068	0.021
	20–30	11.40	4.42	0.117	0.046
2006					
Discharge	0–10	10.27	0.84	0.098	0.011
	10–20	9.58	0.68	0.142	0.008
	20–30	10.55	1.13	0.125	0.022
Control	0–10	8.89	0.94	0.051	0.037
	10–20	8.60	0.68	0.064	0.014
	20–30	9.75	0.88	0.069	0.011

4.5. Net accretion

Three lines of evidence all point to considerable net accretion in the discharge zone. Peat probe surveys were repeated, and showed increased thicknesses. Water surface elevations were tracked as a function of time, and showed steady increases. Topographic surveys were repeated, and showed increases in soil surface elevations.

Peat probing was done in the pre-project year 1977, and repeated in 1998 for a 350 m transect along the discharge line near the tee (Fig. 4). The peat thickness in that area was 1.11 ± 0.22 m in 1977, and 1.53 ± 0.24 m in 1998. The clay underlayer was presumably not changed, and hence there was an increase of 0.42 m of new material in this centrally located portion of the discharge zone. The accretion rate was 2.0 cm/y over this 21-year period of irrigation. This zone would presumably have experienced the greatest accretion. This method of gauging accretion was affected by the presence of the floating mat, which confounds the definition of soil elevation.

Water surface elevations were monitored monthly throughout the project history, during the irrigation season. The staff gages were driven down into mineral soil, and surveyed to establish a datum for each. Typically, the water surface was flat in the backgradient portions of the irrigation area, and sloped away from the discharge line in the downgradient direction. Three gauging stations were located in the backgradient pool area. Hydrological studies showed that irrigation caused a water level rise of 5–10 cm, with buildup and recession occurring in about a day upon pump turn-on or shut-off (Hammer and Kadlec, 1986). Water levels after pump shut-off at the end of the operating season were therefore indicative of the soil elevation of the irrigation area, with variations due to the relative proportions of rain and evapotranspiration during this post-shutdown period. These end-of season water levels displayed a linear increasing time trend (Fig. 5), with a slope of 0.47 cm/y. The increase over 30 years was 14.1 cm. This method of gauging accretion reflects a wide-area spatial average, because water was able to drain around the mound of maximum accretion that was centered on the tee.

4.5.1. Pre-project surveys

The entire 700-ha peatland was surveyed during 1972–1973, on a 400 m grid, referenced to a benchmark on the Deadhorse Dam concrete structure. This elevation was 345.74 m above mean sea

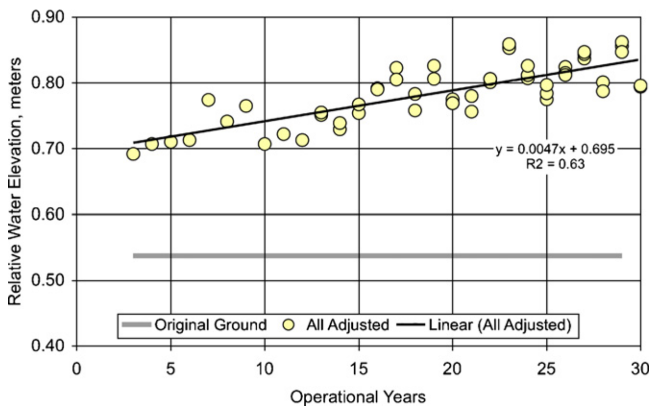

Fig. 5. Water elevation change at the end of the pumping season over the period of record. Measurements are for one to three stations located 220–380 m backgradient from the discharge line.

level, which was assigned a comparative elevation of 4.8 cm. Stations in the modern-day irrigation area were reported to be at about elevation 30.1 cm in 1972–1973. This survey was done in the winter, and the elevations represent the frozen surface. In the irrigation zone, that was a surface of frozen soil, but at other locations, it represented the ice cover on deeper pockets.

An area of about 200 ha was surveyed as a precursor to the wastewater addition project, in 1977. That survey was also done in frozen conditions, and would represent the same soil/ice elevations. The intent was to ascertain the slopes and elevations for positioning the discharge pipeline. There were several stations in the modern-day irrigation area, which have an average elevation of 40.8 cm (Deadhorse Dam = 4.8 cm). While these two surveys do not agree perfectly, they report the winter wetland surface elevation at 35 ± 5 cm. The modern-day irrigation area was surveyed again in 1979, and ground elevations were found to average 40.8 cm (Fig. 6). These elevations are probably comparable to the pre-project surveys, because not much water had been added in the startup year of 1978. If these three surveys are averaged without prejudice to any one of them, the mean wetland elevation in the irrigation area was 37 ± 6 cm prior to wastewater effects.

4.5.2. Post-project surveys

A small region of the irrigation area, along a centerline from 400 m backgradient of the tee, to 100 m downstream of the tee, was resurveyed in 1988. The average wetland surface elevation was

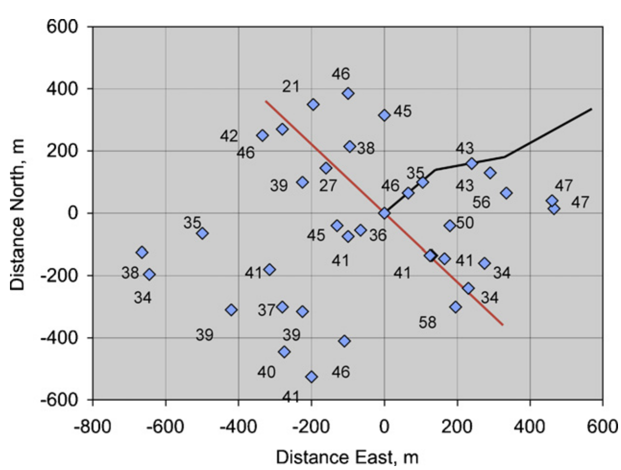

Fig. 6. Pre-project survey. Relative elevations in cm.

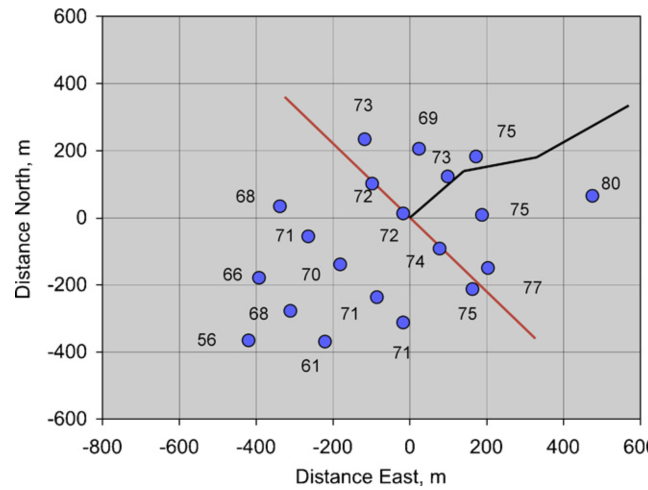

Fig. 7. 2005 survey. Relative elevations in cm.

found to be 59 ± 8 cm, about 22 cm higher than the best estimate of the pre-project level. Most of the stations were within 100 m of the discharge, and therefore the annual average increase of 2.0 cm/y represents the most highly impacted area of the irrigation zone.

A spatial topographical survey was done in winter 2005, covering about 60 ha central to the irrigation area. Twenty stations were located precisely, and marked with steel pipes driven into mineral soil (Fig. 7). Modern satellite techniques were used to measure the elevations of the frozen marsh surface, with five replicates at each location. The Deadhorse Dam reference elevation was also used in this survey, and again assigned the relative elevation of 4.8 cm. The marsh surface elevations shown in Fig. 7 average 70.9 ± 5.5 cm. Over the 27 years of operation, the annual average increase was 36 cm, or 1.33 cm/y, which represents the entire impacted area of the irrigation zone. This estimate of the long-term accretion is affected by the inherent uncertainty in survey methods, especially those employed prior to the 2005 satellite technique. In all cases, the definition of the soil surface was the frozen soil horizon, which may have depended to some degree on the moisture/standing water conditions of the preceding autumn season.

4.6. Surface solids mass balances

Eq. (6) was solved and calibrated, and results in dual exponential behavior along the flow direction (Fig. 8). The first portion shows a change from the incoming TSS to the higher concentrations associated with fertilization. The second portion shows a change from the high fertilization values back down to lower background concentrations associated with the downgradient unimpacted zones. At any location, there is an increasing time trend in the amount of material collected in the sedimentation cups.

Calibration of this model for the sedge-willow cover type utilized the forcing variables depth (15 cm), flow rate (50 m/d) and incoming TSS (20 mg/L). The fertilized and unfertilized background concentrations were selected as 45 and 4 mg/L respectively, compared to tube measurements of 34 and 7 mg/L respectively. The apparent settling rate coefficient was 0.09 m/d, resulting in combined resuspension and generation of 4 and 0.36 $g\,m^{-2}\,d^{-1}$ for fertilized and unfertilized zones. The fit to cup collector data was in general agreement with data (Fig. 8), but not adequate for detailed quantitative interpretation.

Calibration for the cattail area was slightly different. The forcing variables were depth (20 cm), flow rate (40 m/d) and incoming TSS (20 mg/L). The fertilized and unfertilized background concen-

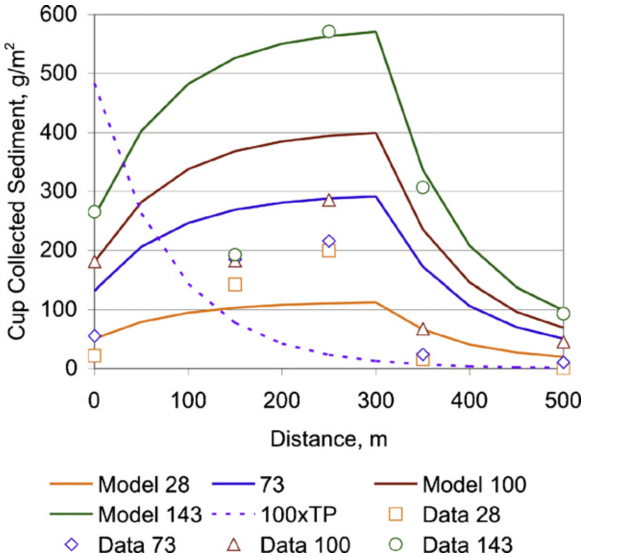

Fig. 8. Settling model results for cup collectors along the flow direction in the sedge-willow cover type for elapsed times of 28, 73, 100 and 143 days. The scaled profile of total phosphorus in the water is shown to illustrate the nutrient-rich zone.

trations were selected as 35 and 6 mg/L respectively, compared to tube measurements of 34 and 7 mg/L respectively. The apparent settling rate coefficient was 0.12 m/d, resulting in combined resuspension and generation of 4.2 and 0.66 g m^{-2} d^{-1} for fertilized and unfertilized zones.

The season totals for the irrigation area provide an approximate idea of the magnitude of solids processes in the system. Over the 30-year POR, the average incoming suspended solids were 21.1 metric tons per year (t/y). The average outgoing TSS was 3.2 t/y, based on an estimated outgoing TSS of 7 mg/L. The mass removal was 85%. However, based on an estimated gross deposition rate of 3.08 g m^{-2} d^{-1} (Table 1), during 143 d, on the 100-ha irrigation zone, the seasonal amount of settling was 440 t/y, or 25 times the apparent removal rate of 17.9 t/y. The amount of generation plus resuspension was 422 t/y. Thus, the solids processing in the system was dominated by internal cycling.

The removed incoming TSS together with the generated TSS that settles, form the input to the wetland sediment pool. However, that sum is not the net accretion for the wetland, because some of the new materials are subject to decomposition.

5. Soil and sediment chemistry

5.1. The root zone

The root zone is perceived to be the region in which biogeochemical processes act to alter the chemistry of wetland water. The primary focus of this study was nutrient (nitrogen and phosphorus) processing, but that in turn was expected to involve other major constituents, namely calcium, magnesium, potassium, iron and aluminum. The pre-project total concentrations of these were determined for the upper soil layers (Table 3). Total iron and aluminum were not determined for the pre-project condition, but later measurements suggest iron was comparable in concentration to calcium, and that aluminum was an order of magnitude less. Processing of phosphorus is generally considered to be more dependent on the amounts of extractable metals than the total amounts. Accordingly, 1 M HCl extraction was used to determine extractable Ca, Mg, Al, and Fe (USEPA method 200.7, carried out by the Soil Science Lab of the University of Florida). Because the additions to the soils of the root zone involve the top sediments, these were also analyzed (Table 4). In general, the amounts of exchangeable metals were comparable in the upper 18 cm, although slightly higher in the discharge zone. However, the mobile sediments on top of the soil column contained higher amounts of exchangeable Ca, Mg, Al, and Fe. Pre-project extractions were conducted using the Bray-2 method (Richardson and Marshall, 1986), and showed comparable amounts of extractable metals, with somewhat less calcium and more aluminum.

Table 3
Major constituents of the antecedent peat in the sedge-willow area of the Porter Ranch peatland. These are %DW for digested solids. Pre-project data are for the top 20 cm of cores taken in the natural wetland (Richardson and Marshall, 1986). Pilot control data are for the top 15 cm of soil in the same area, and away from the pilot discharge (Tilton and Kadlec, 1979).

	1973 Pre-project	1976 Pilot control
Nitrogen	1.7–2.7	2.25 ± 0.18
Phosphorus	0.05–0.07	0.083 ± 0.015
Calcium	1.0–1.7	1.28 ± 0.22
Magnesium	0.12–0.16	0.14 ± 0.01
Potassium	0.05–0.09	0.08 ± 0.01

Of most interest for this project were the amounts of nitrogen and phosphorus contained in the soils of the root zone. Total concentrations displayed a marked effect of the wastewater addition (Table 5, also see Table 3). The levels of both nitrogen and phosphorus remained stable at pre-project levels in control zones distant from the discharge, at about 2.5%DW for nitrogen, and about 800 mg/kg for phosphorus (0.08%DW). In the discharge zone, nitrogen increased slightly, to about 3.0%DW, but phosphorus increased three-fold to about 2500 mg/kg.

The processing of soil samples in this study involved removal of large roots and rhizomes, but dead and fine roots were not removed. Therefore, the overall increased concentrations of N and P in the root zone are likely to be the result of several processes, involving the generation of nutrient-rich root necromass, sorption of porewater phosphorus and nitrogen, and the (yearly) deposits of sedimentary material on the top of the soil column. The origin of new top sediments may in turn be the result of accretions of both macrophytes detritus and bacterial algal and microbial detritus.

Table 4
Amounts of HCl-exchangeable constituents in top soils and new sediments. Cores CP are in the impact zone; cores SC are in the control zone. There were duplicate cores, seven slices per core. There were 10 replicates of suspendable material samples. Units are mg/kg.

	SC1 and SC2 (top 18 cm) 1989		CP1 and CP2 (top 18 cm) 1989		Impact zone suspendable 1988	
	Mean	S.D.	Mean	S.D.	Mean	S.D.
Total P	624	85	2,249	541	2,549	910
Exch P	189	46	847	505	1,278	863
Exch Ca	16,135	2452	25,357	4540	30,084	3980
Exch Mg	1,084	172	1,780	269	2,324	841
Exch Fe	3,274	1942	3,356	654	8,806	7074
Exch Al	664	518	521	327	847	483

Table 5
Nutrient concentrations in the top 30 cm of soils in discharge and control zones, 1978–2006. Means and standard deviations are across multiple sections of multiple cores in each time category. The control zones are located backgradient of the irrigation area, and discharge zone cores came from the central portion of the irrigation area. Numbers of core sections were 13 in 1978–1980, 14 in 1987, 12 in 1989, and 18 in 2006.

	Discharge		Control	
	Mean	S.D.	Mean	S.D.
Nitrogen (%DW)				
1978–1980	1.80	0.44	2.08	0.69
1987	3.12	0.25	2.81	0.38
2006	3.13	0.33	2.30	0.26
Phosphorus (mg/kg)				
1978–1980	930	36	990	19
1989	2249	680	624	84
2006	2552	802	897	142

The carbon content of the original, pre-project peats was 32 ± 10%DW (Richardson and Marshall, 1986) for the top 20 cm. In 2006, the carbon content was found to be 45 ± 2%DW in a control zone, and be 35 ± 1%DW in the discharge zone, for the top 30 cm. This accords with higher ash content in the discharge zone, discussed below.

5.2. Sorption of nutrients

The phosphorus sorption isotherm was determined in two independent investigations. Hammer and Kadlec (1980) determined $K = 313$ and $n = 0.45$ for the sedge peat from the Houghton Lake wetland. This result was later confirmed by Richardson and Marshall (1986), with $K = 334$ and $n = 0.46$ for the applicable concentration range.

The average incoming water TP concentration was 3.5 mg/L for the POR. Although the proportion of dissolved phosphorus was 2.6 mg/L, an upper limit is here considered, for which the entire TP is presumed available for sorption. The corresponding sorbed phosphorus was 550 mg/kg. As a frame of reference, the top 30 cm of soils had a solids content of about 10% by weight, and a somewhat lesser bulk density. Therefore the top 30 cm had approximately 30 kg m^{-2} of dry soil. The sorbed phosphorus would be 16.2 g P m^{-2} at the discharge, and reduce to zero with increasing distance from the discharge, as water concentrations become lower. The amount of phosphorus stored can be estimated from the water concentration profiles, measured in each year, and the isotherm in Eq. (7), by integration over the impacted area. As the water phase concentration front expanded over the first years, the equilibrium sorbed phosphorus also would have displayed a moving front (Fig. 9). However, the sorption potential would have become exhausted in about 5 years. The total amount of sorption-stored phosphorus increased to 2.9 ± 0.7 t, over the last 25 years of the project. This is a computed number, based on a vertical sorption zone of 30 cm, and therefore should be considered only as an approximation. Nonetheless, this storage may be compared to the 56 t of phosphorus added to the wetland over 30 years, of which 53 t were retained. Therefore, an estimated 5% of the storage was as sorbed phosphorus.

Oxidized nitrogen forms, nitrite and nitrate, do not bind to solid substrates, and laboratory studies confirmed that this was the case for the peats at Houghton Lake. But ammonia is capable of sorption to both organic and inorganic substrates, because the ammonium ion is subject to cation exchange. Ionized ammonia may therefore be removed from water through exchange with detritus, organic and inorganic sediments in wetlands. The adsorbed ammonia is bound loosely to the substrate and can be released easily when water chemistry conditions change. At a given ammonia concentration in the water column, a fixed amount of ammonia is adsorbed to and saturates the available attachment sites.

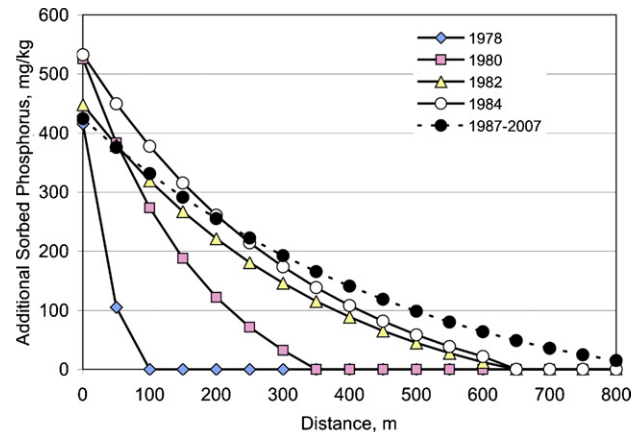

Fig. 9. Advance of the phosphorus sorption front. Points and lines are calculated from the sorption isotherm and water phase concentrations.

The Freundlich isotherm was also determined for ammonium on the Houghton Lake peat and Hammer and Kadlec (1981) determined $K = 16$ and $n = 0.81$ for the sedge peat. The average incoming water ammonium nitrogen concentration was 6.3 mg/L for the POR. The corresponding sorbed ammonium was 70 mg/kg. As for phosphorus, a few years were required to saturate the ammonium sorption potential. The average total amount of sorption-stored ammonium increased to 173 ± 767 kg, over the last 25 years of the project. This is a computed number, based on a vertical sorption zone of 30 cm, and therefore should be considered only as an approximation. Nonetheless, this storage may be compared to the 124 t of ammonium nitrogen added to the wetland over 30 years, of which 121 t were retained or dissipated. Therefore, less than 1% of the ammonium reduction was estimated as sorbed ammonium.

5.3. Vertical concentration profiles

Accretion of new sediments and soils was a major storage mechanism in the wetland, and therefore it is of interest to understand the concentrations of nutrients and other chemicals in the new deposits, and how these compare to the original conditions in the peatland. In most cases, vertical profiles of concentration were found. In general, the amount of chlorophyll in the vertical profile was high near the top, and the degree of humification (pyrophosphate index) was low at the top. Chlorophyll decreased to zero, and humification increased several-fold at the bottom of the root zone.

5.3.1. Ash characteristics

Suspendable material was presumably the source of the new soil layers that accumulated over the period of irrigation. Suspendable material from the irrigation area had an ash content of 31 ± 7% (1980).

Vertical profiles of ash content in soil cores showed low content in surficial zones in control areas, with 10–15% ash in the root zone, which extended approximately 30 cm deep (Fig. 10). Below the root zone, the transition to the underlying mineral soils was evident, with high values of ash content. In 1987, 10 years of irrigation, the ash content of the upper soil horizons in the discharge zone was considerably higher than in control areas, ca. 30% in the top centimeters. By 2006, after 29 years of irrigation, the entire root zone (0–30 cm) was at 30% ash, while control zone profiles remained at much lower levels.

Analysis of the ash on the scanning electron microscope showed that the dominant metals in the ash were calcium, silicon and iron;

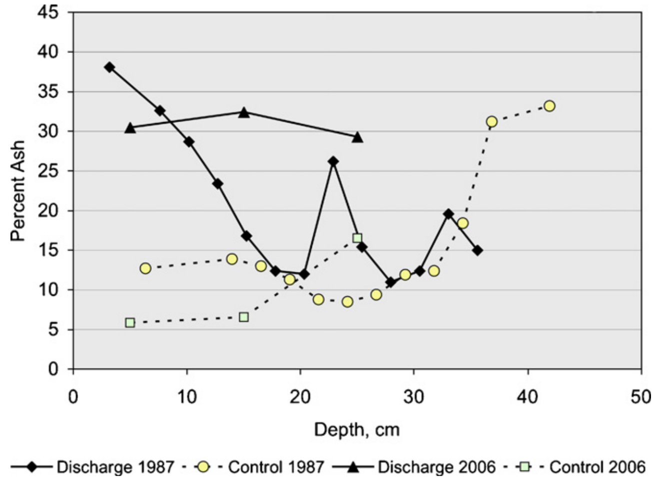

Fig. 10. Ash content of soil profiles for the discharge area and for backgradient control areas after 10 and 29 years of irrigation.

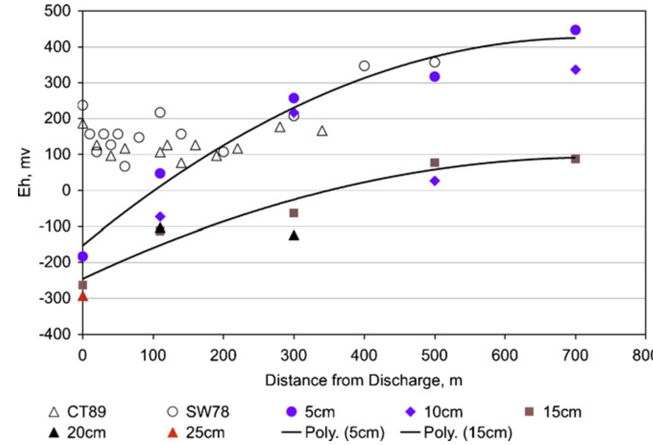

Fig. 12. Redox potential as a function of depth and distance from the discharge at the end of the 1989 season. For comparison, the surface water redox is shown for 1978 and 1989 (open points). Solid points are at various depths in the soil column.

with smaller amounts of magnesium and potassium. There were no pronounced vertical profiles, and no apparent differences between discharge and control zones. Variability between duplicate cores was high.

5.3.2. Ancillary chemicals

The wastewater had considerably higher electrical conductivity (EC) than the antecedent surface waters of the wetland (Kadlec, 2009a). The pre-project level was typically 150–200 μS/cm, and the pumped water contained 600–800 μS/cm. Therefore, it is not surprising that EC was found to be quite high in the upper soil horizons in the discharge zone (Fig. 11). Penetration was only to the bottom of the root zone, even after 10 years of irrigation, indicating that diffusion was very limited, and suggesting that the dissolved materials were brought down into the root zone by the transpiration flux. Much of the elevated conductivity was due to elevated chloride, with the irrigation water in the range of 100–150 mg/L and the wetland background at about 20–30 mg/L.

The redox potential in the pre-project soils was not measured. After 12 years, the redox in the soils near the discharge was lowered considerably, compared to remote locations (Fig. 12). Although surface water was somewhat oxic along the gradient, $100 < Eh < 350$ mv, soils at all depths at the discharge were at $-300 < Eh < -200$ mv. At a distance of 600 m, soils were in the range of $100 < Eh < 400$ mv, with lower values a deeper locations. The negative redox values in the soils at the discharge were an indicator of anoxic and anaerobic processes occurring in those soils. Further elucidation of the potential for the several microbial processes in these Porter Ranch impacted and unimpacted soils may be found in D'Angelo and Reddy (1999).

5.3.3. Nutrients

Pre-project vertical TP profiles were not done. Soil cores were sectioned by depth and analyzed in 1989 (twelfth year of irrigation) and 2006 (twenty-ninth year of irrigation). Phosphorus concentrations were strongly variable with depth at both times in the discharge area, but less so in control areas (Fig. 13). In 1989, the floating mat had not yet developed, whereas it was fully floating at the discharge in 2006. The profiles of TP at the discharge and control zones are very similar in the 2 years. The profile at the edge of the floating mat in 2006 was intermediate between those in discharge and control zones. The top soil layers near the discharge had double or triple the TP concentration of the control area.

Nitrogen profiles were done in 1987 (tenth year of irrigation) and in 2006. Total nitrogen concentrations varied little with depth (Fig. 14). In 1987, concentrations are nearly the same in discharge and control zones. There is more difference by zone in 2006, and

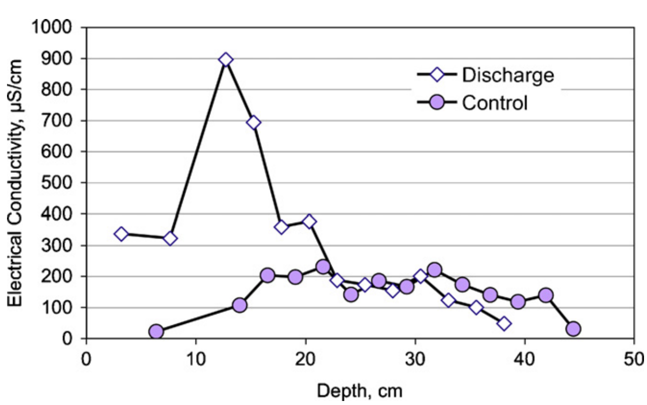

Fig. 11. Electrical conductivity profiles of soil cores in the discharge and control areas in 1987. The surface water EC values were 190–300 μS/cm in the control zone, and 500–600 in the discharge zone.

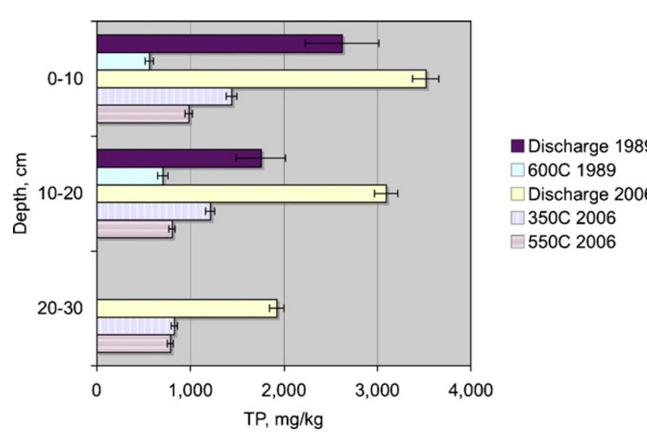

Fig. 13. Vertical profiles of total phosphorus in duplicate cores in discharge and control areas in 1989 and 2006. The cores at 550C and 600C were in a backgradient control zone. The cores at 350C were near the fringe of the floating mat, but in a region of attached roots. The discharge cores were in the floating mat, and fully penetrated the thickness, which was 29.8 ± 1.7 cm. Error bars are standard deviations for triplicate cores. There was no floating mat in the discharge zone in 1989.

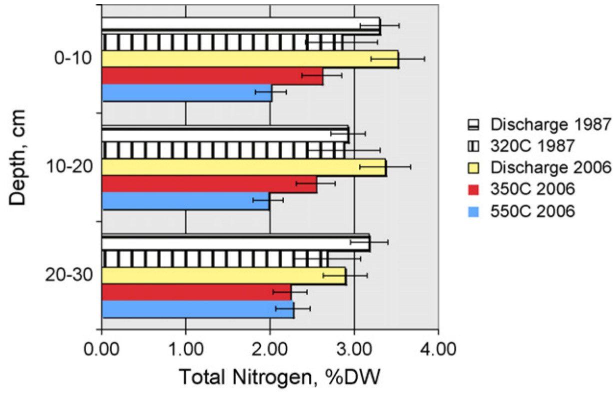

Fig. 14. Vertical profile of total nitrogen in cores in discharge and control areas in 1987 and 2006. The cores at 550C and 600C were in a backgradient control zone. The cores at 350C were near the fringe of the floating mat, but in a region of attached roots. The discharge cores were in the floating mat, and fully penetrated the thickness, which was 29.8 ± 1.7 cm. Error bars are standard deviations for triplicate cores (2006); and for quadruplicate slices in each section (1987). There was no floating mat in the discharge zone in 1987.

a slightly more pronounced profile in the discharge area. However, the effect of irrigation on the nitrogen profiles is not as great as for phosphorus.

6. Discussion

6.1. Character of sediments and soils

The character of the underwater solids changed in the irrigation zone. Prior to wastewater addition, it was fairly easy to distinguish three layers: water over litter over peat. The water layer contained virtually no solid material: 10–30 mg/L if undisturbed, 0.1–0.3% if the sediments were stirred up. The litter layer was situated on a reasonably well-defined peat surface, i.e., it was "rakeable", and consisted primarily of debris from the macrophytes. The litter thickness was not well defined, but was on the order of a few centimeters. Within the peat, a live root zone existed with biomass decreasing to zero at a depth of about 30 cm.

The situation in the irrigation zone is modified from the original in several ways. The water column there contains greatly increased solids, approximately 5% by weight. The consistency (and color) are similar to pea soup. The litter layer still exists, but is now mixed with micro detritus from the water. This is, in large measure, dead algae and dead duckweed. Macrophyte roots have diminished in quantity and moved to higher horizons. As a result, the upper peat layers have greatly reduced strength, since there are fewer live fibers. The dead roots have not consolidated, probably due to the new water regime, which provides little or no chance for drying.

One result of these changes is greatly impeded foot travel; another is the failure of previous soil sampling methods. The original wetland soil permitted tubes to be pushed into the peat, followed by capping and extraction with a peat core. After a few years' irrigation, such cores flowed out of the tube upon extraction. A substitute freezing technique was developed, which worked quite successfully.

The extended hydroperiod and slightly deeper water regime fostered the development of a floating vegetative mat. This *Typha* mat area expanded over the years, and now occupies a large fraction, approximately 30 ha, of the total affected area. Approximately 27 ha of floating mat were found in 2002. Beneath the erect plants, and their attached dry dead parts, is a moist layer of litter. This wet litter layer is comprised of decomposing leaves plus particulate matter. The principal part of the mat consists of a 30-cm layer of roots, rhizomes and sedimentary material. This root mat is tightly woven, both by interlaced rhizomes and by a secondary lacing of roots. The particulate mat solids presumably originate from several sources, including dead algae, fungi, invertebrates and bacteria; plus highly fragmented and decomposed leaves. There was no standing water on top of the mat. Below this root mat, there was a layer of free water, typically 20 cm in thickness near the discharge. Beneath the sub-mat water was a zone of soils and sediments, presumably comprised of antecedent peat together with detritus sloughing from the mat bottom. This soil layer is about 60–80 cm thick in the discharge zone. Beneath this organic layer is the clay that forms the basin for the peatland.

6.2. Sedimentation and accretion

The accumulation of new sediments and soils was considerable over the 30-year project period. This time span was long enough to allow the measurement of accretion via topographical surveys, but there were also the measured phenomena of water level increase and peat thickness increase. The amount of accumulation was highest at the discharge line, amounting to about 2.0 cm/y, as indicated by peat soundings. Broader area topographical surveys indicated that the spatial rate, averaged over about 60 ha, was 1.33 cm/y. Averaged over 30 years, the annual accretion, at a solids density of 0.1 g/cm^3, was 800 t/y. The annual average total suspended solids (TSS) loading to the wetland was 21.1 t/y, and the TSS exported from the irrigation zone was 3.2 t/y. The amount removed, 17.9 t/y, accounts for only about 2% of the observed accretion (Table 6).

The vertical downward gross sedimentation flux was measured with cup collectors, and found to be 440 t/y in the cattail cover type in the irrigation area. If a stationary (unchanging) state existed, then 422 t/y originated from internal sources, including generation of new particulate material and resuspension of settled solids. In control areas, with a presumed low level of generation, the internal sources added to 0.3–0.6 g m^{-2} d^{-1} (Table 1), or 40 t/y on 60 ha. As a limiting case, all of this may be attributed to resuspension, and presumably was similar in the irrigation area. By difference, the generation of new sediments would be 382 t/y. When added to removal of incoming TSS, the sedimentation would have been 400 t/y, or about half the observed accretion.

The balance of the accreted material presumably originated from the above-ground macrophyte necromass, and from the death of roots in the shallow soils, with the 400 t/y corresponding to 667 g DW/m^2. The total end-of season standing crop of above- and below-ground macrophyte biomass in the irrigation area was about 6000 g DW/m^2 (Kadlec, 2009b). Therefore, about 11% of the end-of-season biomass would need to end up as an undecomposed residual. Unfortunately, typical litter decomposition studies, including those for the Houghton Lake wetland (Kadlec, 2009b) do not involve roots, nor do any of the published studies determine the undecomposed residual. Therefore, the quantities described here must be regarded as rough estimates.

The accumulation of 1.3 cm/y of material in the irrigation zone is entirely commensurate with observations at other treatment wetlands (see for instance, Reddy et al., 1993; Craft and Richardson, 1993; Rybczyk et al., 2002; Kadlec and Wallace, 2008). This rate of increase has interesting long-term consequences. The manner of accretion has sometimes been presumed to be sequential vertical layering (Rybczyk et al., 2002; Kadlec and Walker, 1999), but that view is likely to be overly simplified. At least two factors argue against simple layering: vertical mixing of the top soils and sediments (Robbins, 1986), and the injection of accreted root and rhizome residuals at several vertical positions in the root zone. Nonetheless, new residuals are deposited on the wetland soil sur-

Table 6
Period of record (POR) loads of nitrogen and phosphorus to and from the 100 ha irrigation area. Loads of sediments are for the 60 ha study area, determined from sediment cup studies and the accompanying mass balance model. Nutrient storages are estimated from the biomachine model (Kadlec, 1997, 2009a). Quantities in italics are estimated from other data sources (see Kadlec, 2009a). Note that metric tons per 100 ha is the same as grams per square meter.

	Water (pumping season average) 1000 m^3/y	Sediments (30 years total, 60 ha) metric tons	Phosphorus (30 years total, 100 ha) metric tons	Nitrogen (30 years total, 100 ha) metric tons
Inflow	599	630	56.43	135
Rain	341		0.58	13.2
Organic				89.9
Fixation				391.5
Resuspension		1,200		
Generation/water		11,460		
Generation/plants		12,000		
Total	940	25,290	57	629.6
Outflow	470	90	0.85	1
Evapotranspiration	470			
Organic				38.5
Settling/water		13,200		
Accretion/plant		12,000		
Biomass			7.68	27.5
Sorption			2.91	0.2
Burial			45.6	562.4
Total	940	25,290	57	629.6

face, from various sources. The most easily visualized is the litterfall of macrophyte leaves, which results in top deposits of accreted material after decomposition. One observation is that treatment wetlands may be expected to fill up with new solids, and provision may be required to hydraulically accommodate the rise in soil level, although that was not necessary at Houghton Lake. But perhaps of more importance is that such accumulations are very unlikely to be spatially uniform. Local topographic highs and lows may develop, and produce relatively steep local surface elevation gradients. There was some evidence of this at Houghton Lake, because the rise in water elevation (0.47 cm/y) was considerably less than the rise in land elevation (1.33 cm/y average). This suggests that water was following topographic lows, and avoiding topographic highs. This phenomenon is elsewhere called channeling, and is known to be inimical to efficient treatment (Kadlec and Wallace, 2008).

Only one municipal wastewater polishing wetland has been serviced for solids removal: the Orlando, Florida Easterly Wetland inlet cells (White et al., 2004). The removal of accumulations restored good hydraulic patterns, and restored original water quality performance. It was suspected that uneven accumulations of new sediments were affecting flow patterns, and reducing efficiency (Sees, 2005). The inlet nine percent of the wetland was excavated 45 cm, after 15 years of operation. Although accretions were not measured, they were a major fraction of the 45 cm excavated. The over-excavation restored more than the original freeboard, and resulted in a great improvement in hydraulic efficiency.

A significant portion (27 ha) of the impact zone developed into a floating cattail mat. The mat region was still the minority of the irrigation zone, however. It is unlikely that the sediment processes are the same as those in areas with surface water but no mat. The under-mat environment is extremely hostile to algae, because of light limitation. Incoming algal materials were undoubtedly killed and filtered in a small zone near the discharge. It is probable that the undersurface of the root mat sloughs particulates, which fall to the bottom of the water layer. Importantly, there was no duckweed–algae component of the ecosystem in the mat region. There was no surface water to support it, and no light to foster it under the mat. There was undoubtedly a microbial community under the mat that contributed to treatment, but it would have been different from that in surface water. Water re-emerging from under the mat would have had to pass through the fringe of the mat, thus preventing downstream transport of suspended material. The generation of macrophytes litter decomposition products continued in the mat region, but the surface water generation of microdetritus was absent. Although there were surely generation processes under the mat, mediated by bacteria and possibly fungi, it is doubtful that these would have been of the same magnitude as surface processes. Consequently, the parameterization of the sedimentation model (Eq. (2)) should be different than for the non-mat situation. It may be speculated that the loss of the duckweed–algae cycling on sediment generation and nutrient removal would be detrimental to treatment. Some degree of compensation might be expected, due to the fact that some of the new sedimentary material is deposited on the bottom of the under-mat water, and thus is removed from the influences of surficial processes. This is important in other treatment situations, in which toxic materials are involved. Under-mat deposition takes such chemicals out of the reach of surface feeder organisms (Smith and Kalin, 2002).

6.3. Nutrient storage

Water-borne nutrients interact strongly with wetland vegetation and associated biota, which provide both short-term and sustainable long-term storage of this nutrient. Soil sorption may provide initial removal, but this partly reversible storage eventually becomes saturated. Over the POR, less than 3% of the phosphorus and less than one percent of the nitrogen were removed to sorption storage (Table 6). Uptake by biota, including bacteria, algae, and duckweed, as well as cattails, forms an initial removal mechanism. Maximum standing crops were achieved only in areas immediately adjacent to the discharge line. As has been shown, strong gradients existed, with biomass decreasing in the flow direction. When these gradients are considered, via the calibrated biomachine model, the average vegetative storages in the entire irrigation area were 7.7 g P m^{-2} and 27.5 g N m^{-2} (Table 6).

Cycling through growth, death and decomposition returns most of the biotic uptake, but an important residual contributes to long-term accretion in newly formed sediments and soils. Such accretion was the dominant mechanism at Houghton Lake. The expansion of the wetland vegetative community, and its associated nutrient storages, have been described in Kadlec (2009b). Despite the apparent complexity of these several removal mechanisms, data analysis for this site has shown that relatively simple equations can describe the sustainable processes (Kadlec, 1997).

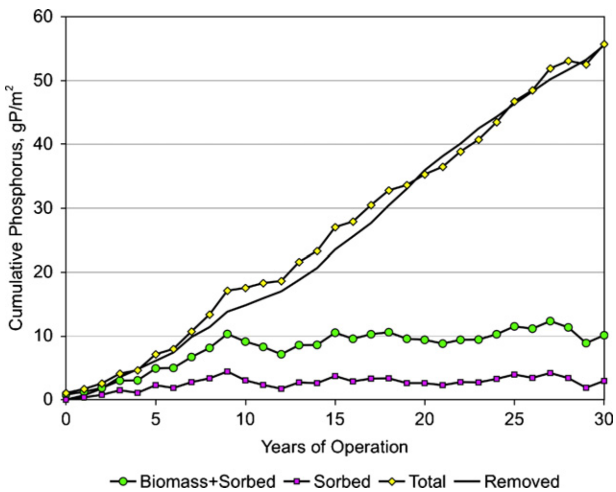

Fig. 15. Phosphorus storages as a function of time.

Sustainable removal of phosphorus is to burial in the form of new accretions of soils and sediments. The residuals from the biomass cycling include both microdetritus and macrophyte litter. The former is the object of the TSS-sedimentation model (Eq. (6)), and the supporting cup collection and agitation experiments. The combination of accretion due to both microdetritus and macrophyte litter has been described in Kadlec (2009b). Biomass growth and the resulting generation of residuals, with their nitrogen and phosphorus content, has been described by a "biomachine" model (Kadlec, 1997). This is a simple model, in that it assigns a portion of each year's primary production to become undecomposed residuals. But, the growth of vegetation is presumed to depend on the nutrient status of the water, which was a measured spatially variable in each project year. This buildup is additional to the amount of N and P needed to saturate wetland sorption sites.

When water of a given concentration is added to a substrate that has sorption capacity, sorption occurs until the entire soil of the wetland is loaded to the solid phase concentration corresponding to that water concentration. This time period was relatively short for the Houghton Lake wetland, and was in part determined by the simultaneous utilization of nutrients for buildup of the new crop of biomass. The calculated time progression (Fig. 15), using the biomachine model, shows that the transient phenomena (sorption and grow-in) were essentially complete in under 10 years. After the establishment of a nearly fully developed new biomass with its accompanying nutrient cycle, there was a relatively steady buildup of accreted materials (Fig. 15). By the end of the 30-year period, this new soil storage was dominant (Table 6), accounting for 81% of the phosphorus input, and 73% of the nitrogen input.

The accretion fluxes estimated from the biomachine model were quantitatively corroborated by measurements of the amounts of new soils and their N and P content. The average accretion of 800 t/y (800 g m^{-2} y^{-1}) of solids (Table 6) carried an average concentration of 1900 mg P/kg (Table 5), resulting in a 30-year storage of 45 t of P (1.5 g P m^{-2} y^{-1}), which corresponds to the model-calculated removal to accretion of 45.6 t of P (Table 6). This close correspondence is the result of calibration of the model, in which the fraction of the biomass standing crop that is undecomposed was an adjusted parameter at 23%. The average P concentration of the depositing material was reasonable well-known, from numerous measurements of the vegetation and the settling solids (0.27%).

Other investigations have similarly found that the phosphorus removed from entering waters may be found in the new soil accumulations in the wetland. The amount of such accretion has been quantified in a few instances for free water surface wetlands (Reddy et al., 1993; Craft and Richardson, 1993; Rybczyk et al., 2002). In a study of P removals in a Florida wetland, Walker (1995) found that the P removed from the water over a 30-year period was found in the new soil accretions.

The amount of nitrogen in the accreted material was much higher. The average accretion of 800 t/y (800 g m^{-2} y^{-1}) of solids (Table 6) carried an average concentration of 2.68%DW (Table 5), resulting in a 30-year storage of 643 t of N (21.4 g N m^{-2} y^{-1}). The average N concentration of the depositing material was not as well-known, but was a combination of biomass at up to 2.4%DW, litter at up to 3.6%DW, and sediments at up to 4.1%DW. A mid-range value was assumed, of 3.6%DW. The biomachine model then showed a 30-year removal of 562 t of N (18.7 g N m^{-2} y^{-1}). Notably, it was presumed that fixation provided a significant amount of nitrogen, about 62% of the input. This possibility is discussed further in Kadlec (2009a). This assumption is based on the amount of nitrogen found in accretion, which considerably exceeded that entering with rain and wastewater. It may also have been possible that some nitrogen was extracted from antecedent soils and re-deposited in the newly formed soils in the top layer, especially in the mat zone.

7. Conclusions

The Houghton Lake wetland effectively removed a large fraction of the incoming TSS, approximately 80%. However, the internal cycling of solid particulates inside the wetland was much, much larger. Sediment cups trapped large amounts of material, and the measured amounts of suspendable materials on the wetland bottom were also very large. Therefore, particulate processes were dominated by generation and cycling of internally generated sediments. The origins of the generated particulates appeared to be roughly equal proportions from micro-flora and micro-fauna, compared to above and below ground macrophyte detritus.

Sediment generation and accumulation resulted in a slow buildup of the soils in the impacted area of the wetland. The amount of such accretion was measurable via topographic surveys, because of the long duration of the project. Spatial variability and inherent inaccuracies of surveying techniques limit the ability to utilize this procedure for short-term projects. The soil level increases resulted in water level increases, and these too were measurable over the long-term. The site had a very dense clay under-basin, which facilitated peat thickness measurements, and these also confirmed the buildup of new soils. One result of increased soil surface increases was the implied effect on internal wetland hydraulics. The newly created internal surface gradients, although irregular, were of the same magnitude as the original land slope. Without water level management, it is possible that the wetland surface rise will eventually create land out of water, and such less than 100% hydroperiods could foster the release of nutrients via the oxidative decay.

The initial configuration of the wetland was shallow-water over litter and peat, with emergent plants dominated by sedges. The water column quickly developed dense growths of algae and duckweed, both biomass and necromass, which in total produced a semi-mobile floc that occupied most of the water column. The canopy changed to a near monoculture of *Typha latifolia*. However, in later years, a floating cattail mat developed, which had an essentially dry surface. Water flowed underneath the mat, in a region of altered sedimentary processes, which were not quantified or well understood. The mat contained an enhanced amount of mineral material, for which there is no obvious explanation.

This study indicates that the long-term storages of nutrients in a lightly loaded natural wetland are dominated by the formation and accretion of the new soils. Although some storage also occurred in the development of new, larger standing crops of vegetation, that amount was important only in the first few years of operation. Phosphorus storage due to sorption, although quantifiable, was likewise not important after a few years. Thus, after a startup period of approximately 5 years, virtually all of the added phosphorus was stored in new soils and sediments. The fundamental predictive presumption that sorption would dominate P removal at this site (Richardson and Marshall, 1986) has been proven false.

Although some wetland science literature correctly identifies accretion as a principal long-term storage for phosphorus (e.g., Craft and Richardson, 1993; Reddy et al., 1993; Rybczyk et al., 2002), much of the treatment wetland literature does not directly address the soil-building mechanism as a primary route of phosphorus or nitrogen immobilization. For instance, USEPA (1999) states: "New constructed and natural wetlands are capable of adsorbing phosphorus (P) loadings until the capacity of the soils and new plant growth are saturated." Mitsch and Gosselink (2000) identify only chemical precipitation, adsorption and plant uptake as removal mechanisms. Crites et al. (2006) state: "Adsorption and precipitation reactions are the major pathways for phosphorus removal. . ." The Houghton Lake wetland data demonstrates that these authors, and many others, have overlooked the fundamental process of soil accretion for P storage. Interestingly, USEPA (2000) correctly identifies the sustainable mechanism as "accretion and burial."

This oversight has further consequences. It is well-known that iron and aluminum are strong correlates for the sorption of phosphorus (Lijklema, 1977; Reddy et al., 1993, 1998). It is also well-known that the oxidation state of iron is important, with lesser binding capacity in the reduced form. Redox potentials in the new soil horizons in the impacted zone were quite low, and would suggest minimal binding capacity. It is probable that the minimal sorption present at Houghton Lake would be so influenced, but not the burial of phosphorus incorporated in undecomposable tissues of macro- and micro-flora and fauna. The conclusion of P release due to lowered redox is not necessarily valid for necromass phosphorus.

Nitrogen removal poses a different difficulty in the literature. Many treatment wetlands are exposed to heavy nitrogen loadings, although a substantial fraction are lightly loaded, as was the Houghton Lake system. If heavily loaded, then the (proper) conclusion is that plant uptake and burial is of minor importance (USEPA, 2000; Crites et al., 2006). Microbial processes of nitrification and denitrification have ample "left-over" nitrogen with which to work. Consequently, some authors have made the overly broad statement that "Nitrogen removal in constructed wetlands is accomplished by nitrification and denitrification." (Crites and Tchobanoglous, 1998). That is not true for those treatment wetlands that operate with less-than-adequate nitrogen supplies, as was the case at Houghton Lake. There was inadequate incoming nitrogen to support the creation of undecomposable residuals. In this situation, it is likely that the wetland found the required nitrogen via the mechanism of fixation. Any nitrification and denitrification would have added to the requirement for fixation to provide the nitrogen needed for biomass and sediment creation.

In the long-term, the surficial soils of the wetland became those generated by the new biogeochemical cycle. Antecedent soils were located beneath the new root zone, and were not necessarily in a position to interact strongly with water and biota. Because they were out of touch and possibly modified, interpretation of long-term behavior should not be based upon the antecedent soil conditions in a wetland.

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