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Wastewater treatment at the Houghton Lake wetland: Vegetation response

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ABSTRACT

This paper describes the vegetation 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, during which large changes in the plant communities occurred. An average of 600,000 m^3 $year^{-1}$ of treated wastewater was discharged seasonally (May 1–October 31) 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 (DIN). 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. Phosphorus was stored in new biomass, increased soil sorption, and accretion of new soils and sediments, with accretion being dominant. The irrigation area underwent large changes in ecosystem structure, in which the original plant communities were displaced by *Typha* spp. There was an initial fertilizer response, characterized by much larger standing crops of vegetation, at about triple the crop in control areas. Increased biomass was accompanied by increases in tissue nitrogen and phosphorus content, by factors of two and three, respectively. The plant community shift, from the initial sedge-willow and leatherleaf-bog birch cover types to a cattail-dominant cover type, progressed to a 83-ha area over the 30-year period of record (POR). The interior portion of this new cattail patch became a floating mat. There were large gradients in stem densities and stem heights within the impacted area. The response times of the vegetative community shifts were on the order of 10 years for 63% of the final impact zone development. The grow-in time for development of a new larger standing crop in the discharge zone was also 10 years. The impacted area was stable at the 30-year time, without any further moving fronts. Around the cattail zone, there were fringe areas that contained a mixture of the original cover types intruded by relatively small amounts of cattail.

1. Introduction

Details of the project history and other aspects of the project have been described in Kadlec (2009-a). 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

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summer half-year to a smaller pond, and thence to the Porter Ranch Wetlands, an existing natural peatland 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 (Haag, 1979). 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. The hydraulic loading was approximately 600,000 m^3/year, applied during the pumping season, which averaged 128 days in length, during May–October. The effluent contained 3.5 ± 2.0 mg/L of total phosphorus (TP), and 7.5 ± 5.2 mg/L of dissolved inorganic nitrogen (DIN = $NH_4N + NO_XN$; 20% NO_XN). The annualized rate of TP addition was 1.87 gP/m^2 year, and 0.11 gP/m^2 year was exported from the irrigation zone. The irrigation area removed a total of 53 metric tons of phosphorus over the 30-year period of record (POR). The average annual loading of dissolved inorganic nitrogen to the irrigation zone was 4.50 gN/m^2 year, and only 0.11 gN/m^2 year left the irrigation zone. A total of 131 metric tons of dissolved inorganic nitrogen 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 (2009-a).

The water addition was about double the amount of rainfall experienced by the irrigation zone. The hydraulic loading was approximately 0.6 m/year. The accompanying annualized nutrient loads were 4.5 gDIN/m^2 year and 1.9 gP/m^2 year. 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 vegetation within the irrigation zone was affected by the extra water and nutrients. Additionally, some areas adjacent to the irrigation zone experienced changes that may or may not have been associated with the addition of the treated wastewater. The remainder of the peatland did not experience such changes. It is the purpose of this paper to document the changes in the Porter Ranch peatland vegetation that occurred as a consequence of the addition of the treated wastewater.

2. Methods

A number of procedures were utilized to assess vegetation 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).

Species composition was determined by identifying species presence and the relative abundance of plants on 86 plots, each 32 m^2 (4 m × 8 m). These plots were initially located at 10 m intervals on two transects parallel to flow, ranging from 100 m backgradient, to 200 m downgradient. After 1990, when it was apparent that the zone of impact was larger than anticipated, the plots were relocated to span the entire irrigation area and beyond, from 400 m backgradient to 1100 m downgradient. Ten cover categories were used for this visual estimation method: (1) <1%, (2) 1%, (3) 2–5%, (4) 6–10%, (5) 11–25%, (6) 26–33%, (7) 34–50%, (8) 51–75%, (9) 76–90%, (10) 91–100%. These surveys were conducted by Professor F.B. Bevis and his assistants from Grand Valley State University.

Aboveground plant biomass was determined by harvesting random triplicated plots in the discharge, backgradient and control areas. Aboveground parts were clipped from triplicate 0.25 m^2 areas in late August during each project year, and at other times in selected other years. The categories of material in the discharge cattail area were live, standing dead (not prone in the water), and litter. The categories in the sedge areas were above ground live plus standing dead and litter. The backgradient areas contained a mixture of these two categories. This litter layer may be removed by gentle raking with the fingers, without any tearing of roots or dead leaves. It occupies a vertical thickness of about 10–20 cm. The plant parts were field weighed, and subsamples of each were oven dried to constant weight (24–72 h at 80 °C), in order to develop a correlation between oven dry weight and field weight. In some years, below ground roots and rhizomes were harvested, from lesser areas, typically 0.1 m^2. These were rinsed, dried and weighed. Duckweed sampling was conducted using an isolation ring. 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 0.112 m^2 of surface water, vegetation and suspendable material. Duckweed (*Lemna*, spp.) was gently skimmed from the interior using a wire mesh strainer. Replicate samples were dried and weighed in the laboratory.

Vertically stratified clips were conducted to ascertain the volumes and surface areas of the under-water plant parts. Clipping was accomplished with the aid of an enclosing wire frame with rings at established elevations. Triplicate clips were performed for patches of *Typha latifolia* and *Typha angustifolia*, at three times during the season, all at the discharge line. The densest patches of plants were sought for clipping, thus providing an upper bound for material density, although single plant apportionment was later conducted to translate the results to other stem densities. Plant material was collected from 0 cm to 10 cm, 10 cm to 20 cm, and 20 cm to 30 cm elevations above the root-rhizome mat. The frame was pushed down into the water until the bottom ring could not be pushed further, with about 20 kg pressure. Live and dead plants were enumerated, and the biomass returned to the laboratory. Each aliquot of clipped material was then immersed in water until fully saturated, drained, and weighed. The material was then dried to constant weight at 80 °C, and weighed again. The wet weight was presumed to correspond to a bulk density of approximately 1.0 g/cm^3, and therefore the volume of the material was calculable.

Starting after a few years of wastewater discharge, portions of the cattail area near the discharge became detached from the soils and formed a floating mat. In 2002, the area of this zone was estimated by probing the cattail root zone along three pipeline transects, at 10 m intervals. Further "probing" was done downgradient of the discharge line, via an Argonaut™ All Terrain Vehicle (ATV) expedition. The condition of the root mat – anchored or floating – was assessed by pushing with a pole from the pipeline support dock. If the root mat could be visibly moved with the pole, a floating condition was recorded. On the ATV expedition, the assessment involved visual observation of the mat upon stopping the vehicle. In regions of floating mat, stoppage caused a wave to propagate through the neighboring cattail stand. Plugs were cut from the mat by use of a sharp machete. Rectangular plugs, approximately 20 cm × 20 cm in planar area, were excised and removed to ascertain the mat structure. Such plugs weighed about 15 kg (30 lb) and could be removed by hand. The vertical thickness was about 20–25 cm, comprised of live and dead rhizomes and roots, plus adhering soil and sediment particles. The plugs were washed thoroughly in the wastewater discharge of a wide open gate in the distribution pipeline (flowing at about 160 m^3/d). This process removed all but live and strongly attached dead root and rhizome material. The wet weight of the washed roots and rhizomes was then determined by field weighing, and a subsample returned for dry weight determination. Triplicate mat cores were taken.

Litter decomposition was studied in 1983–1985 (Chamie, 1976; Chamie and Richardson, 1978). Vegetation from seven categories was used in this study: sedge leaves (*Carex* spp.), willow leaves and stems (*Salix* spp.), bog birch leaves and stems (*Betula pumila*), leatherleaf stems (*Chamaedaphne calyculata*), and cattail leaves (*Typha latifolia*). Approximately 200 (18 cm × 28 cm) bags were made from nylon netting with 1 mm × 1 mm openings. This opening size and bag size represents a compromise between possible fragmentation loss and freedom of movement of decomposer organisms. The bags contained approximately 15 g dry leaves or 30 g dry stems. A set of subsamples were weighed and reserved for oven drying to determine a field weight-oven dry weight correlation. These litter bags were placed on June 22, 1983 (wood), and September 2, 1983 (leaves), in an irrigation (boardwalk) area, and in control areas (+800 m, −600 m from the discharge line. All bags were firmly set in the existing litter layer: above the peat and below water. Bags were harvested on 6 dates (5 for wood) spanning 766 days (838 for wood). They were immediately frozen at the field laboratory, to preserve them for later analysis. The thawed bags were thoroughly rinsed to remove adhering algae and duckweed (Lemna spp.), and all intrusive roots and stems were picked out. The bags were then oven dried at 85 °C to constant weight (24–72 h), and weighed to the nearest 0.01 g. The empty bags were reweighed as a check on the filling and weighing procedure.

The nutrient content of selected samples of the various vegetation compartments was determined. Dried ground biomass was wet-ashed and then subjected to analysis for total phosphorus and total nitrogen. Early analyses were conducted in the laboratory at the University of Michigan, while later in the project, analyses were performed in the laboratories of the Department of Soil Science at the University of Florida. Dried samples were analyzed for a large suite of elements via neutron activation analysis at the University of Michigan during early project years. Samples were exposed to the University of Michigan cobalt radiation source, and the energy spectra of emitted neutrons was then determined after a short and long periods. Comparison to NBS vegetation standards provided the necessary calibration. The sulfur content of selected vegetation and soil samples was determined at the University of Michigan using a Leco™ Sulfur Analyzer.

Aerial photography was used to determine the extent of alteration of the peatland in the vicinity of the discharge. Color, false color infrared and multi-spectral digital imaging were used across the period of record, in 25 of the 30 years. About half the years involved vertical scale-calibrated imagery, while the other half involved only hand-held, oblique photography. The monthly field expeditions for sample collection provided the means of ground-truthing of photo interpretation.

3. Species composition

The plant associations of the peatland were greatly altered in the vicinity of the discharge, but not at distances remote from the effects of the water and nutrients. The resulting new ecotones, or gradients in species abundance and composition, are important features of the altered wetland.

3.1. Zonation

The peatland irrigation site originally contained areas of two distinct vegetation types: one with predominantly sedges (*Carex* spp.), willows (*Salix* spp.) and bog birch (*Betula pumila*); and a second with leatherleaf (*Chamaedaphnae calyculata*) and bog birch (*Betula pumila*). These accounted for 68% and 19% of the whole peatland, respectively (Wentz, 1975). Isolated small patches of cattail (*Typha latifolia*) were also present, accounting for 1.8% of the cover. However, virtually all of this cattail cover type was located in or near the discharge zone. The area near the discharge, which more than encompassed the affected area, comprised 200 ha (500 acres) as shown in Fig. 1A. The principal plant species in each cover type were measured in 1973. The relative dominance of the various plants species on plots was evaluated for the entire wetland, as indicated in Table 1. Cattail was dominant in 12.6 ha, in locations that subsequently were upgradient of the discharge pipeline (Wentz, 1975). The wetland contained small aspen islands (*Populus tremuloides*), accounting for 2.5% of the cover overall, but was also mostly located in the 200 ha area near the discharge, where it comprised 18 ha, or about 9% of the cover. The edge of the peatland contained alder (*Alnus rugosa*) (3.4% of the cover), and willow, which graded into surrounding upland areas containing primarily aspen.

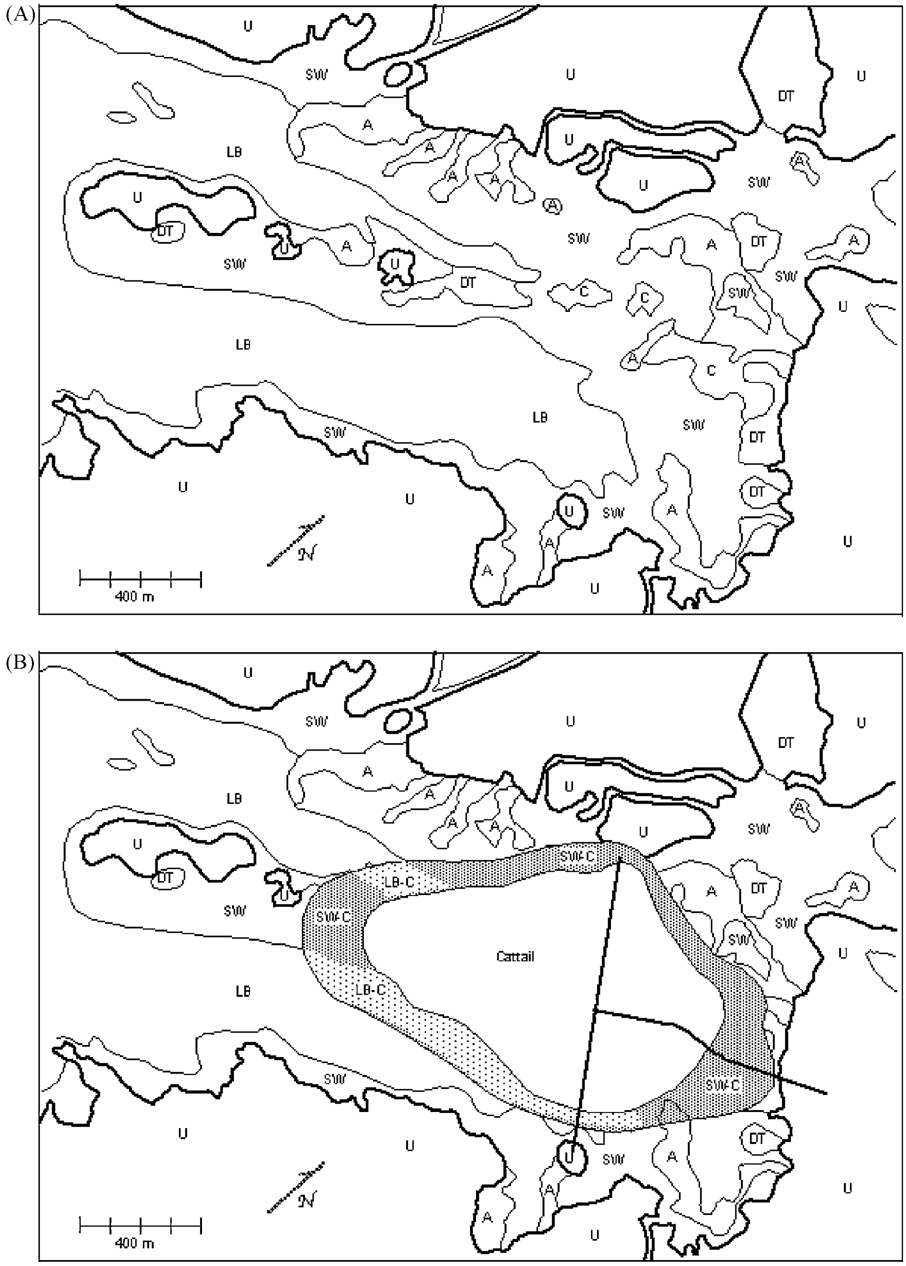

Fig. 1 – Cover maps of the Porter Ranch peatland in its original condition (upper panel A) and after 30 years of irrigation with treated wastewater (lower panel B). In the upper panel A = Alder; C = Cattail; DT = Dead timber; LB = Leatherleaf + Bog birch; C = Cattail; SW = Sedge-willow; U = Upland. Additionally in the lower panel LB-C = Leatherleaf + Bog birch + Cattail; SW-C = Sedge-willow + Cattail.

The addition of treated wastewater created a heavily impacted region, centered on the discharge pipeline (Fig. 1B). This zone, of about 85 ha extent in the 30th year of the project, became strongly dominated by *Typha* spp. Around this central zone was a band of approximately 100–200 m width, in which there was a gradient of *Typha* density, from the dense interior cattail cover to a very sparse occurrence of cattail intruding upon the original vegetation assemblages. These ecotone zones, or fringe zones, comprised approximately an additional 50 ha. Intrusion occurred in areas formerly populated with sedges, willows, bog birch and leatherleaf. *Typha* stem densities were about 50–70 m^{-2} in the dense zone (Fig. 2), and heights were over 2 m. The drop in the relative abundance of cattail was steep at both backgradient and downgradient locations. Although stem densities and heights were very much lower in the fringe zones, the cattail was visually obvious as an intruder. Its presence was easily noted at densities 100 times lower than in the heavily impacted zone. Fig. 3 shows the original sedge community and the replacement cattail community that developed during the project.

Aerial photography was used to delineate the area of greatest impact in 25 of the 30 years. Regions of dense cattail were

Table 1 – Percent frequency of occurrence of plant species pre-project (1973), and during the first 15 years of operation.

		1973 162[a]	1980 90[a]	1981 90[a]	1982 90[a]	1983 90[a]	1984 90[a]	1985 90[a]	1992 86[a]
Salix spp.	Willow	72	96	96	96	96	94	90	12
Calamagrostis canadensis	Blue-joint reedgrass	64	100	90	90	88	88	84	45
Cicuta bulbifera	Water hemlock	16	85	82	81	80	78	76	6
Carex spp.	Narrow leaved sedges[b]	82	77	77	77	77	77	71	14
Carex spp.	Broad leaved sedges[c]	32	15	15	15	14	13	13	23
Spiraea alba	Narrow-leaf meadow-sweet	66	73	74	73	73	71	68	7
Betula pumila	Bog birch	70	65	65	65	65	65	65	72
Galium trifidum	Small bedstraw	P	65	65	62	60	58	59	0
Lycopus spp.	Bugleweed	19	62	60	59	59	56	51	2
Ferns	Ferns	25	62	63	63	63	62	59	65
Typha latifolia	Wide leaf cattail	17	46	49	52	59	62	66	99
Aster junciformis	Aster	82	54	53	51	54	51	51	5
Lemna minor	Duckweed	5	19	24	29	43	49	54	99
Chamaedaphne calyculata	Leatherleaf	60	23	24	26	26	27	27	35
Muhlenbergia glomerata	Marsh muhly	49	23	23	24	23	23	21	4
Iris virginica	Virginia blue flag	19	12	14	15	17	19	22	5
Utricularia vulgaris	Bladderwort	10	12	14	15	17	17	17	2
Lysimachia terrestris	Swamp loosestrife	16	15	15	15	15	15	15	9
Equisetum fluviatile	Water horsetail		12	14	15	15	15	15	5
Solidago gigantea	Goldenrod	55	12	13	13	13	13	11	1
Scirpus cyperinus	Wool grass	15	0	0	2	4	4	6	0
Mosses	Mosses	P	85	83	79	77	73	68	52
Sphagnum spp.	Sphagnum	P	12	0	0	0	0	2	7

P indicates present, not estimated.
[a] Number of plots.
[b] Includes *C. lasiocarpa*, *C. aquatilis*, *C. oligosperma* and others.
[c] Includes *C. rostrata*, *C. comosa*, *C. lacustris* and others.

able to be clearly demarcated on color and false-color photos due to elevated reflectance in wavelengths corresponding to elevated chlorophyll levels. Such imagery showed duckweed as light colored, and circles of *Typha angustifolia* as darker shades (see Fig. 4). This visually affected area grew at a decreasing rate throughout the period of record (POR) of the project (Fig. 5). An exponential model describes the time trend in the affected area:

$$A = A_{max}(1 - \exp(-t/b)) \quad (1)$$

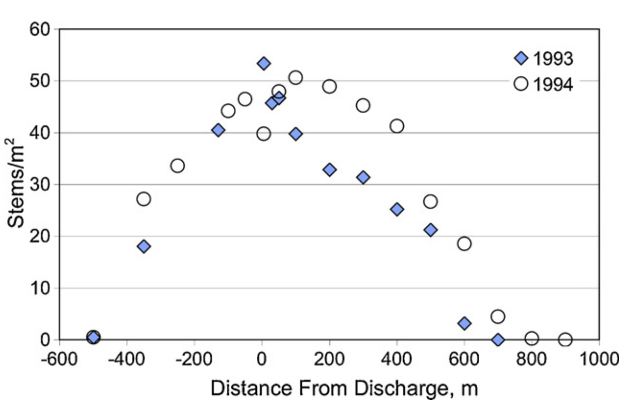

Fig. 2 – Cattail stem densities along a gradient parallel to flow in 1993 and 1994. Points are means of triplicates, with S.D. typically less than 10%.

where A = affected wetland area (ha); A_{max} = maximum affected wetland area (ha); b = time constant for area expansion constant (year^{-1}); t = time since discharge began (year).

This model fits the time trend (Fig. 5) with an $R^2 = 0.91$. The maximum area was projected to be $A_{max} = 83$ ha. The model time constant for areal expansion, during which 63.2% of the expansion has taken place, was $b = 9.0$ years.

3.2. *Phytosociology*

During one pre-project year and nine of the project years (1980–1985, 1992–1994), the relative abundance of plant species on plots in different cover types was determined. The original zone of study was from 100 m backgradient to 200 m downgradient, with the intent to study the entire impact zone and surroundings. However, by 1992 it was apparent that this zone was completely occupied by the dense cattail community, with near-complete disappearance of some sedges, willow, bedstraw, water hemlock, *Spiraea* and other species (see Fig. 3). Cattail and duckweed were totally dominant by that time (Table 1). Accordingly, study plots were relocated into three types of cover: (1) the original sedge and leatherleaf-bog birch zones of the wetland as controls, (2) the heavily impacted zone near the discharge pipeline, and (3) the fringe zones that contained mixtures of original and replacement vegetation. The relative abundance of the major components (greater than one percent of the cover) of the vegetative canopy are given in Table 2 for the period 1992–1994. The fringe and discharge zones are distinctive, in that these contain considerable pro-

Table 2 – Relative abundance of plant species on plots in different cover types in 1992–1994.

Scientific	Common	Control zones		Fringe zones			Discharge zone
				Backgradient	Downgradient		
		Sedge BB	LL BB	Sedge	CT BB	CT LL	CT
		13.5[a] 20[b]	24.6[a] 10[b]	15.2[a] 5[b]	7.2[a] 31[b]	13.6[a] 31[b]	3.7[a] 19[b]
Sphagnum spp., etc.	Mosses & Lichens	28	30	21			
Carex lasiocarpa	Woolly fruit sedge	82	15	44			
Carex aquatilus	Water sedge	20	36	1			
Salix spp.	Willow		31	8			
Spiraea alba	Narrow-leaf meadow-sweet		47				
Utricularia vulgaris	Bladderwort	33					
Drosera rotunifolia	Roundleaf sundew	23					
Acer rubrum	Red maple		20				
Vaccinium macropcarpon	Large cranberry	36					
Betula pumila	Bog birch	44	50		34	69	
Chamaedaphne calyculata	Leatherleaf	32	88			29	
Typha latifolia	Wide leaf cattail			54	87	38	100
Lemna minor	Duckweed				22	38	25
Calamagrostis canadensis	Blue-joint reedgrass					13	
Galium trifidum	Small bedstraw			6	8		1
Thelypteris palustris	Fern			6	4	5	

BB = Bog birch; LL = Leatherleaf; CT = Cattail.

[a] Species/plot.

[b] Number of plots.

portions of *Typha*, and the control zones do not. However, it should be noted that the selected control areas were not in the locations of the original *Typha* communities.

The number of species per plot, a measure of diversity, was low in the discharge area (3.7), compared to both fringe and control areas (7.2–24.6). In total, the number of species found on control plots (42) was somewhat greater than the total found on fringe plots (32) or in the discharge zone (32). These major dominance numbers do not reflect the numerous species that occurred in very low numbers in the original peatland, and are now absent in the discharge zone. These included *Saracenia purpurea* (pitcher plant), *Andromeda glaucophylla* (bog rosemary), *Chelone glabra* (turtlehead), *Asclepias incarnata* (swamp milkweed), *Viola palustris* (marsh violet), *Rosa palustris* (swamp rose), *Campanula aparinoides* (marsh bellflower), *Rumex verticillatus* (dock), *Eriophorum* spp. (cottongrass) and others.

3.3. *Appearances and disappearances*

The major dominance numbers also do not reflect the invader species that appeared late in the project history. Importantly, the plots did not include any of the patches of *Typha angustifolia*, which increased in abundance over the last 10 years of the irrigation. Further, in the last year of record (2007), there was a large intrusion of nightshade (*Solanum dulcamara*) in the discharge zone, where the vines created a tangle interwoven in the dense stand of *Typha*.

Two small patches of narrow leaved cattail (*Typha angustifolia*) developed early in the project, one about halfway out to the distribution tee, and another along the southern branch of the distribution pipeline. Subsequently, there was an expansion of patches of *Typha angustifolia* into the *Typha latifolia* matrix in the central region of the distribution pipeline (*T. angustifolia*) was not present in the original peatland, and did not appear obviously on aerial photos up through 1984. Aerial photos from 1995 showed only a few circular patches, totaling about 0.2 ha. Aerial photos from 2000 showed these circular patches to be increasing in number to about 30, and growing in size, then totaling about 2.0 ha. Photos from 2005 show an estimated 4 ha of narrow-leaf cattail, and those from 2007 show about 5 ha (see Fig. 4). *T. angustifolia* is clearly visible at ground level as well, because it is about 0.5 m taller than adjacent *T. latifolia*. Stem densities of *T. angustifolia* were about double those of *T. latifolia*, 121 ± 41 versus 71 ± 21.

The most notable disappearance was of woody shrubs and trees. The aspen (*Populus tremuloides*) community near the pipeline tee, which contained 21 live trees in 1979, had completely succumbed in 1983. A second aspen island, located 500 m downgradient, had also totally succumbed by 1984. Significant defoliation occurred in 1980, and to a lesser extent in 1981 for the entire wetland. The aspen on the edges of the peatland were also killed, in backgradient and side locations where the shore slopes gradually. The hydroperiod of the wetland was increased to 100% due to irrigation in the entire shore-to-shore wetland backgradient and to the side of the discharge pipeline, although nutrients did not find their way to these shoreline areas. The alteration of the water regime was the presumed cause of death along much of the wetland perimeter, in a band up to 50 m wide at a few locations. Long-dead timber at these and other wetland locations indicated that similar events may have occurred naturally in the past.

Fig. 3 – The pre-project vegetative cover was a sedge meadow in many locations (upper panel, prior to 1978). The post-project vegetation was dominated by cattail in the vicinity of the discharge (lower panel, 2006).

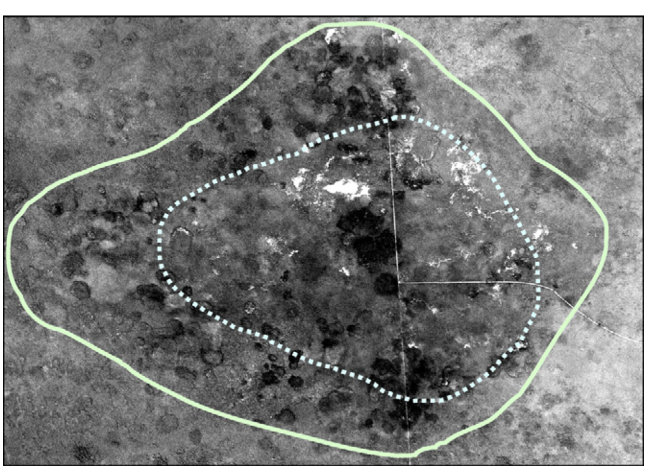

Fig. 4 – Aerial photo of the irrigation area from August 8, 2007. The white areas are open water, with varying degrees of duckweed cover. The black areas are *Typha angustifolia*. The solid outer line indicates the approximate zone of *Typha* dominance. The inner dashed line indicates the approximate extent of the floating mat.

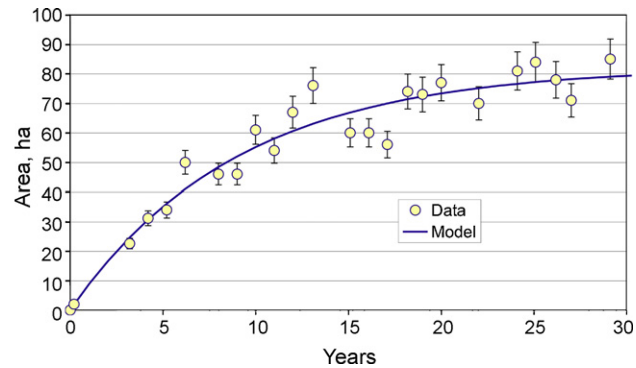

Fig. 5 – The visually affected area as a function of time. Data is from color aerial photography. The fit is to an exponential rise, which calibrates to a final area of 83 ha, at the current irrigation rate. Error bars are estimated from repeated photo interpretations.

The original peatland contained large areas of the sedge-willow community. The willows were typically about 2 m in height, and approximately half the stems were standing dead of indeterminate age (Wentz, 1975). Both live and dead stems were physically unaffected by the presence of the sedges. The number fraction of standing dead stems in the discharge zone went from 52% during pre-project and early 1980s, to 67% in the late 1980s, to 88% in the early 1990s. Thereafter, neither live nor standing dead willows were found in the discharge zone. The (separate) evaluation of phytosociology showed a decline in the presence of *Salix* spp. from 96% in 1980 to 12% in 1992. The demise and the loss of upright standing dead was likely the result of two factors. First, the water line on the stems was subject to insect and possibly fungal attack. Second, the dead stems did not remain standing more than about 1 year in the presence of the tall, dense *Typha*. The dead cattail leaves were forced to the peatland surface by wind and snow, and tore down the entangled standing dead of both willow and bog birch. Bog birch fared better than willow in general, but it too ultimately disappeared near the discharge pipeline, in the zone of densest cattail.

3.4. *The floating mat*

The extended hydroperiod and slightly deeper water regime fostered the development of floating vegetation. During the first few years of irrigation, when sedges were still present near the discharge pipeline, clumps of *Carex* spp. were in a nearly hydroponic mode, as evidenced by the ability to move the clumps across the water/sediment surface. Later, the Typha community also developed into a floating mat. The Typha mat area expanded over the years, and now forms a large fraction of the total affected area (Fig. 4).

The vertical structure of the mat consists of several layers (Fig. 6). The canopy typically consists of live and standing dead cattail leaves and flower stalks, with live leaves to a vertical height of about 3 m, and dry dead leaves to about 1 m. Live stem densities are approximately 70 m^{-2} for *Typha latifolia*, and 120 m^{-2} for *Typha angustifolia*. Beneath the erect plants,

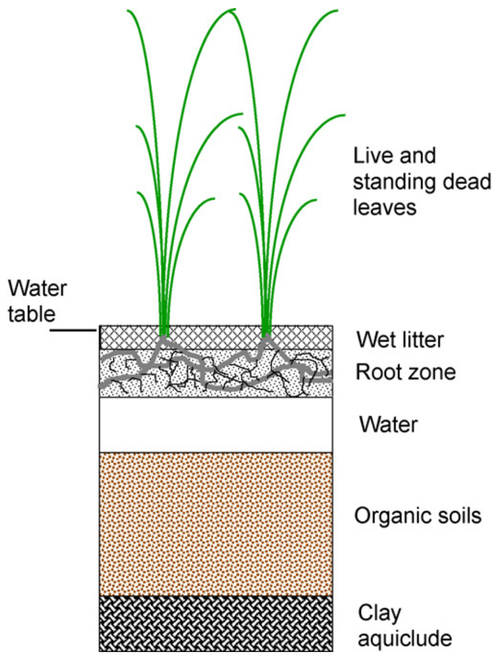

Fig. 6 – Vertical structure of the cattail mat at the discharge.

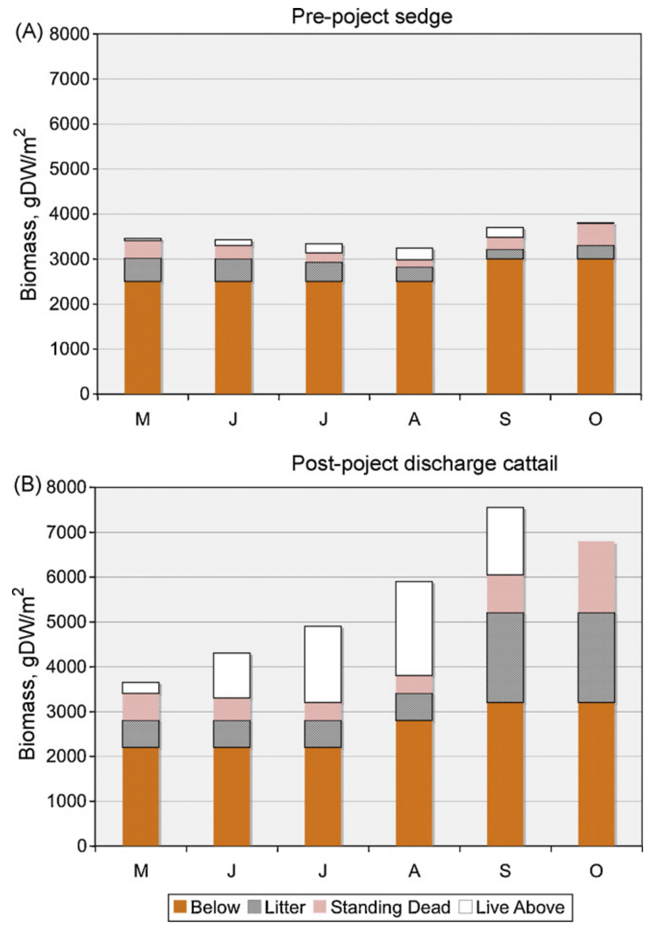

Fig. 7 – Seasonal growth patterns for the original sedges and the replacement cattails. (A) Pre-project sedge. (B) Post-project discharge cattail.

and their attached dry dead parts, is a moist layer of litter. This wet litter layer is comprised of decomposing leaves plus particulate matter. Particulate solids presumably originate from several sources, including dead algae, fungi, invertebrates and bacteria; plus highly fragmented and decomposed leaves. The vertical thickness of the main mat was about 20–25 cm, comprised of live and dead rhizomes and roots, plus adhering soil and sediment particles. Soils and sediments comprised the majority of root zone solids. This root mat is tightly woven, both by interlaced rhizomes and by a secondary lacing of roots. The mat weave is strong enough to support the Argonaut™ ATV carrying four people, with only slight depression, and in most places allows foot travel on the mat without breakthrough.

Below the root zone, there was a layer of free water, typically 20 cm in thickness near the discharge. Standing on the mat caused it to depress through this water layer, and exude that water. 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 is the clay that forms the basin for the peatland. Approximately 27 ha of floating mat were found in 2002 (Fig. 3). There was no standing water on top of the mat. After removal of a plug from the mat, water ponded in the excavation to within 2–4 cm of the top of the litter layer.

The under-mat water layer carried the added wastewater, as evidenced by temperatures. The vertical temperature profile in the discharge (mat) zone showed a maximum in the under-mat water, of about 4–8 °C, corresponding to the warm added water (Kadlec, 2009-c). Temperatures in the mat were lower, presumably due to evaporative cooling. Temperatures below the under-mat water were also cooler, due to heat conduction downward to the cool deep soils.

4. Phytomass

The amount of plant materials was greatly increased by the addition of the treated wastewater. Here the terminology of Mueleman et al. (2002) is used, which suggests that the total is "phytomass," composed of living material (biomass) and dead (necromass). Dead material may be erect (standing dead) or prone (litter). A distinction is also made between above and below ground compartments.

4.1. *Annual growth patterns*

The growing season at Houghton Lake begins approximately at the very end of April, and senescence of aboveground plant parts is typically complete at the end of October. The peak standing crop is typically found at the end of August.

Fig. 7 shows the seasonal pattern of growth and shrinkage of the various phytomass compartments, for the antecedent sedges (Richardson et al., 1976) and the replacement cattails. Note that the below ground biomass does not vary a lot during the year, and the biomasses of roots and rhizomes are not much different for the "before" and "after" cover types. The major difference is in the amount of above ground biomass that is generated each year, with the fertilization causing five

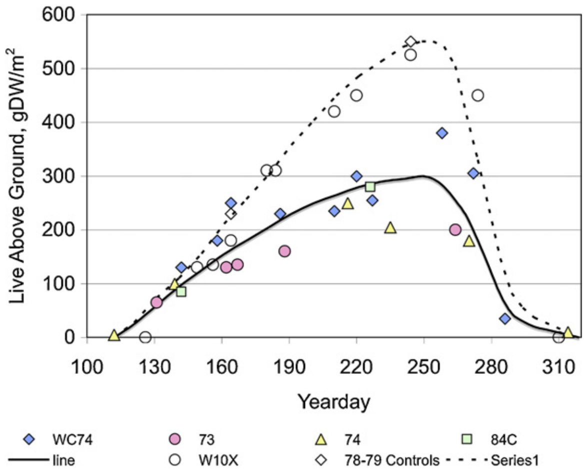

Fig. 8 – Initial surge in the growth of sedges as a result of added water and nutrients. Pre-project field data is labeled 73 and 74. Lab controls are WC74, 1984 field controls are 84C. Discharge areas are labeled 78–79, and lab fertilization is labelled W10X.

to ten times as much emergent material. The backgradient location, which experienced extra water but no excess nutrients, followed a similar pattern; but the peak above ground standing crop of cattail was about half that of the discharge zone.

There was an initial surge in the growth of sedges in the discharge area. The biomass approximately doubled. This effect had been predicted by the laboratory work of Wentz (1975), who fertilized sedge and compared it to controls without fertilization. His "10X" nutrient addition had comparable growth stimulation to that observed during 1978 and 1979 in the field project (Fig. 8). However, these were transitory phenomena only, because the sedge community was totally displaced by cattail.

4.2. *Gradients*

The central impacted zone of the wetland was not uniform in the size and density of vegetation, but rather exhibited gradients that matched the gradients in water quality. The project started with clip plots along the flow direction, during 1978–1983. In general these showed a large decrease in aboveground biomass in the flow direction, with twice as much near the discharge line as 200 m downgradient. For example in 1980, there was 1580 gDW/m^2 at the discharge, versus 790 gDW/m^2 at 200 m downgradient. However, as the project progressed through the years, clip plots became increasingly difficult to accomplish along transects, due to the extreme difficulty of foot travel.

A less satisfying quantification was the stem density of *Typha latifolia*, as shown in Fig. 2. A second, approximate quantification was obtained by measuring cattail plant heights and stem densities. The product of these is a rough relative measure of biomass. Transects were assayed in 1993 and 1994, and showed a decrease of relative biomass with distance from the discharge line, in both the backgradient and downgradient directions (Fig. 9). Plant heights ranged from more than 2 m tall near the discharge to less than 1 m tall further away from the discharge. The principal drop-off in relative biomass occurred in a 200-m wide zone on the leading and trailing edges of the cattail zone.

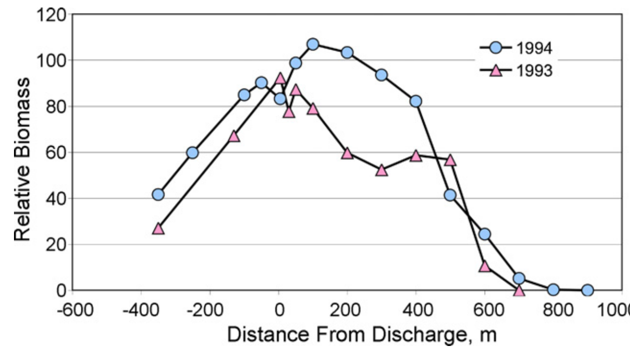

Fig. 9 – The profile of relative biomass of *Typha latifolia* with distance from the discharge line. Relative biomass is stem density times plant height.

A second characteristic of the impacted zones was the presence of duckweed (mostly *Lemna minor*), which occurred in both the dense cattail and in the fringe zones. In patches of open water and in animal trails, large quantities of *Lemna* were present. The maximum density was found near the discharge line, with up to 200 gDW/m^2 (Fig. 10). This quantity was comparable to the live aboveground biomass of the sedges that formerly occupied the region. The seasonality of the duckweed cover was less than that for the emergent plant parts. The *Lemna* plants remained green all winter, under the snow and sometimes imbedded in the ice. A fairly large standing crop was present at the time of spring thaw, and remained prevalent throughout the growing season and into the next autumn and winter. However, the development of the floating mat eliminated duckweed from those locations. Further, dense cattail stands with surface water contained sparser duckweed. It seems probable that these floating plants moved episodically with the water, toward more downstream locations.

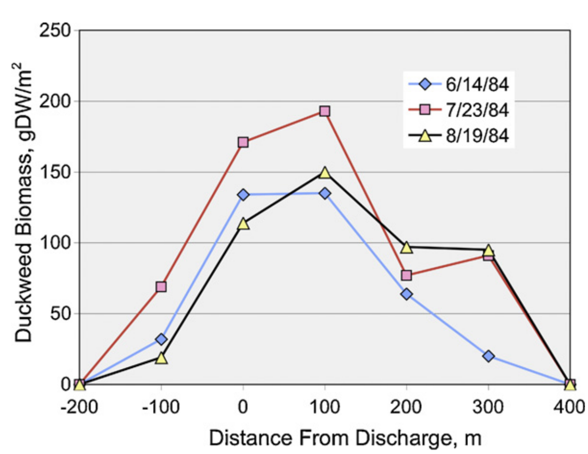

Fig. 10 – The profile of duckweed (*Lemna minor*) with distance from the discharge line.

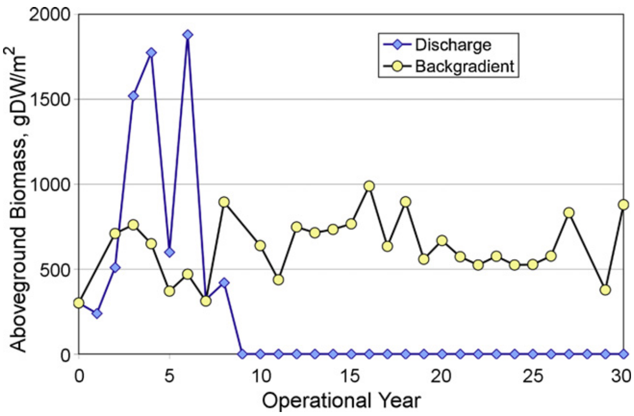

Fig. 11 – End of season standing crop of aboveground live and standing dead biomass of sedges in the area of the discharge and in backgradient areas.

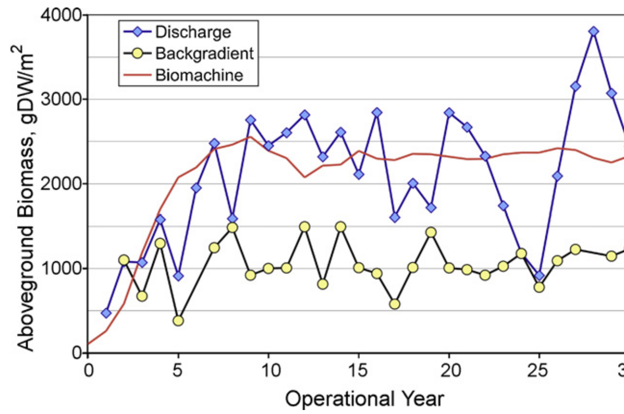

Fig. 12 – End of season standing crop of aboveground live and standing dead biomass of cattail in the area of the discharge and in backgradient areas. The biomachine line represents a logistic model driven by the water quality and quantity applied at the pipeline.

4.3. *End of season crops*

Clip plots were maintained at several locations: at the tee in the discharge line, in the backgradient water-only area, and in control areas that saw neither water nor nutrients. These were conducted at the end of the growing season, in late August, in every year, and therefore provide insights about the long-term behavior of the standing crops. There is now much more biomass in the discharge area than in the backgradient areas. The current backgradient areas are subject to deeper water in summer due to ponding, but not to increased nutrients. These areas have developed sparse, short stands of cattail, interspersed among the sedges. Cattail plants are not canopy dominant in backgradient areas, but are biomass dominant. Visually, cattails are noticeable in almost all backgradient zones.

Fig. 11 shows that sedges in the discharge zone went through a large increase in biomass followed by a complete replacement by cattail in 1986. In the backgradient zone, there exists a variable, enhanced, but now apparently stable, annual peak standing crop of sedge. The average value over the last 28 years was 642 ± 174 gDW/m^2, which is about double that of the sedge areas of the pre-project wetland (Table 3). However, the areal extent of the “pure” sedge community has decreased dramatically. It is probable that cattails were able to displace the sedges by aggressively competing for rooting space, and by developing a taller stand that shaded out plants relegated to the understory.

Fig. 12 shows a 30-year oscillating trend in annual peak standing crop of cattail at the discharge line. There was a 5-year down trend in the above ground biomass of *Typha latifolia* during 1999–2003 That pattern reversed in 2003–2006, with end-of-season standing crop returning to values above those that typified years 6–20 of the project. During 2006–2007, there was another downturn in standing crop. Muskrat herbivory was partially responsible for the low biomass in years 5 and 8, but thereafter these animals were present only in very low numbers, and had no observable impact. There was been no such corresponding pattern in the backgradient control area, where biomass averaged only 40% of that in the discharge area. The amount of aboveground cattail phytomass in the discharge area was very large compared to the pre-project conditions in cattail areas (Table 3).

The end of season standing crops of woody shrubs also underwent a pattern of enhancement followed by a complete “crash” in the discharge area. The pre-project standing crops of stems were 135 gDW/m^2 live and 146 gDW/m^2 dead for willow, and 301 live and 157 dead for bog birch. The peak crops were achieved in 1987–1989, approximately 10 years after commencement of irrigation. These maximum observed standing crops of stems were 478 live and 495 dead for willow, and 1096 live and 256 dead for bog birch. By 1992, there were essentially no remaining clumps of willow or bog birch in the discharge zone.

4.4. *Litter crop and decomposition*

Litter is here defined to be prone dead and decomposing materials, and the associated sediments. It is differentiated from standing dead, which is erect necromass. Litter was harvested from previously clipped plots, and displayed variability depending on what fraction of the previous year’s standing dead had been forced to the surface and into the water. There was about 300–600 gDW/m^2 of litter in control and backgradient areas, and in the pre-project condition. In contrast, there was considerably more litter in the discharge zone cattail community (Table 3), with values up to 3000 gDW/m^2. There was also a time trend upward (Fig. 13), which may have reflected the development of floating mat conditions, as well as a slow rate of decomposition.

Litter decomposition followed a pattern of an initial sharp drop in weight, followed by a steady decline over the next few years (Fig. 14). This may be represented by an exponential with a pre-multiplier:

$$\frac{W}{W_0} = A \exp(-kt) \tag{2}$$

Table 3 – Biomass in vegetation compartments at the end of the growing season (gDW/m^2).

Cover type	Location	Period	Litter	Live	Standing dead	Total above standing	All above	Below
T. latifolia	Discharge	1981–2007	2296	1263	1030	2243	3963	2368
T. angustifolia	Discharge	2001–2002		1773	1236	3008		
T. latifolia	Backgradient	1981–2007	394	665	363	1067	1455	4200
T. latifolia	Control	1986–1994	650	654	483	1153	1864	
T. latifolia	Control	1982–1984	656			1036		
T. latifolia	Pre-project	1973	297	530				
Carex spp.	Discharge	1981–1985	673			999	1452	1300
Carex spp.	Backgradient	1981–2007	436			642	1098	
Carex spp.	Control	1982–1984	445	384				2300
Carex spp.	Pre-project	1973		241				2650

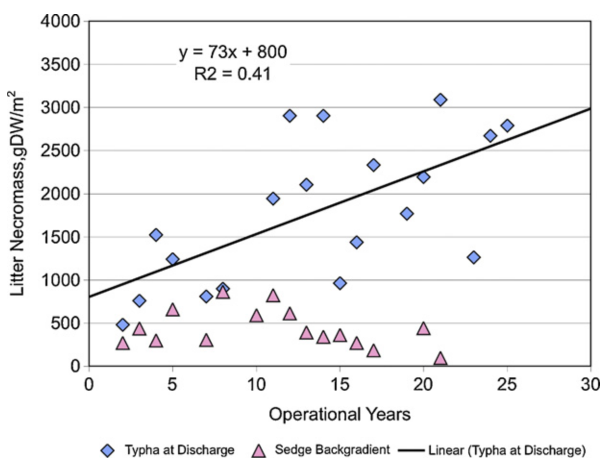

Fig. 13 – End of season standing crop of litter necromass of cattail in the area of the discharge and sedge in backgradient areas.

where A = fractional initial drop in weight; k = decay rate coefficient (year^{-1}); t = time (year); W_0 = initial dry weight (g); W = dry weight at time t (g).

The regressed values for A and k, over time periods of 766 days for leaves and 838 days for wood, are given in Table 4. The half-lives of leaf litter were 2–3 years for sedge, bog birch and willow leaves, where $t_{1/2} = 0.693/k$. These have been termed "second stage" half-lives, because the effect of an initial weight loss has been considered in setting the regression

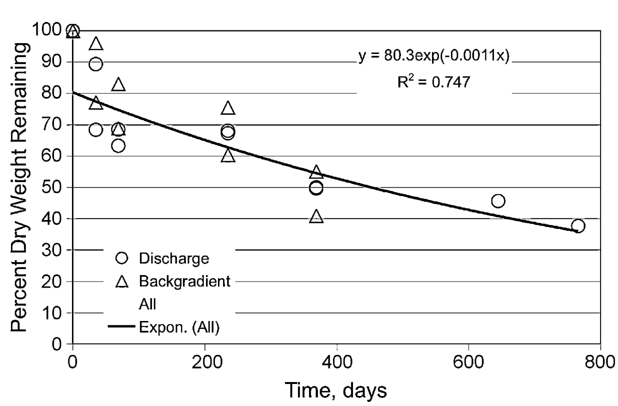

Fig. 14 – Loss of weight of cattail leaf litter samples with time during 1983–1985. The trend line is for the combined data from discharge and backgradient locations.

intercept (Kadlec, 1989). Cattail litter decayed more rapidly, with a half-life of 1–2 years. Wood was more persistent, with half-lives of 5–10 years.

Thirty-three bags could not be found, despite their being attached to nylon string to firm stakes. The principal factor appeared to be string breakage, by animal or human travel. A second contributing factor was the dense, plant and litter cover in the irrigation area, within which it was difficult to find anything. As a result, the intended triplicates became duplicates at each harvest time.

4.5. *Stem densities and areas*

Treatment wetland literature contains numerous references to the volumes and submerged surface areas of plants, as these relate to the "porosity" of the water body, and to the area available for attachment of biofilms that assist in pollutant removal (Crites et al., 2006; USEPA, 1999). The clip plots for this study were intentionally selected to encompass the maximum number of stems within the wire frame. Consequently, the raw data are biased toward extremely high stem densities. Two bases are available to translate the raw data to stem densities that are characteristic of the wetland as a whole: the stem densities in the larger wetland compared to stem densities in these high density plots; and the standing crop biomass in the high density clip plots compared to that in the end of season biomass clip plots. At the end of the growing season, individual plants were harvested, sectioned into 0–10 cm, 10–20 cm, 20–30 cm, and >30 cm length intervals; dried and weighed. This procedure was followed to tie the biomass numbers to the standard end-of-season standing crop measurements that have been performed in all project years.

The volume of plant material was obtained from a fixed height of 10 cm, and consequently the planar cross-sectional area of the material was calculable. This planar cross-sectional area was then divided by the number of plants to obtain a mean cross-sectional area per plant, and an equivalent cylinder diameter (D). The biomass surface area in any aliquot was then calculated as $4/D$ times the volume of material (equivalent cylinder area). Typical water depths at the Porter Ranch treatment wetland were about 10–15 cm, but this data permits determination of volumes and areas at depths up to 30 cm.

The end-of-season standing live biomass for the high density plots was 4300 g/m^2 for *Typha latifolia*, and 8600 g/m^2 *Typha angustifolia*. This may be compared to the standing live biomass of 1200 g/m^2 for *Typha latifolia* from the end of season clip plots. Stem densities in the discharge area

Table 4 – Initial weight losses and decay rate coefficients for litter of several types. Pre-project data is from Chamie and Richardson (1978). Discharge and control experiments were in 1983–1985.

	Discharge		Control		Pre-project	
	A%	k (year^{-1})	A%	k (year^{-1})	A%	k (year^{-1})
Cattail leaves	76.2	0.33	89.5	0.58		
Sedge leaves	80.5	0.37	88.1	0.26	95	0.45
Willow leaves	53.5	0.25	63.0	0.32	85	0.46
Bog birch leaves	54.2	0.21	66.6	0.24	80	0.47
Willow wood	87.5	0.154	85.1	0.009	105	0.182
Bog birch wood	86.8	0.063	104.7	0.217	95	0.082
Leatherleaf wood	107.4	0.061	104.6	0.067	105	0.092

for *Typha latifolia* were measured to be 71 ± 21 stems/m^2 ($N = 16$ years) compared to a late summer/early fall mean of 153 ± 23 stems/m^2 in the high density plots in this study. Stem densities in the discharge area for *Typha angustifolia* were measured to be 265 ± 6 stems/m^2 in the high density plots in this study, compared to counts of 121 ± 41 stems/m^2 ($N = 5$ years). Values in Table 5 were normalized to 70 stems/m^2 for *T. latifolia*, and 140 stems/m^2 for *T. angustifolia*, to provide more representative estimates, however the reader may easily reproportion the areas and volumes to other stem densities. This adjustment is made only for the area and volume descriptions of this section.

There was not much effect of season on volume or area, but in early spring and fall, the plant material was mostly standing dead, while in summer it was standing live. Dead horizontal leaves were always present, but comprised very little of the biomass. The void volume fraction in the first 30 cm of plant height was 95–97% for the high density plots, which translates to 99% at end-of-season on a marsh-wide basis (Table 5).

Table 5 – Surface area and void volume of cattail vegetation. Means of triplicate stratified clips, units are m^2/m^2 for area, and % for void volume. These values have been normalized to 70 stems/m^2 for *T. latifolia*, and 140 stems/m^2 for *T. angustifolia*.

	5/24/97	7/29/97	10/26/97
T. latifolia			
Area			
Bottom	0.95	0.95	0.98
0–10 cm	0.33	0.33	0.22
10–20 cm	0.19	0.24	0.17
20–30 cm	0.14	0.21	0.15
Total 30 cm	1.61	1.73	1.52
Voids			
Total	97.8%	97.3%	98.7%
T. angustifolia			
Area			
Bottom	0.96	0.96	0.97
0–10 cm	0.45	0.41	0.37
10–20 cm	0.30	0.33	0.30
20–30 cm	0.22	0.30	0.30
Total 30 cm	1.93	2.01	1.94
Voids			
Total	97.7%	97.6%	98.0%

The marshwide basis normalizes the data to 70 stems/m^2 for *T. latifolia* and 140 stems/m^2 for *T. angustifolia*. These values contradict the 65–75% range in water volume unoccupied by plant biomass, which is recommended in USEPA (1999, 2000).

The biomass and bottom surface area in the first 30 cm of plant height was 1.7–2.5 m^2/m^2 for the high-density *Typha latifolia* plots, and 3.2–4.9 m^2/m^2 for the high density *Typha angustifolia* plots. These translate to 1.6 m^2/m^2 and 2.0 m^2/m^2 on a marsh-wide basis, respectively (Table 5). These values are comparable to values reported for *Typha* by USEPA (1999). On a mass basis, the equivalent cylinder surface area for the first 30 cm of plant was 1.00 ± 0.25 m^2/kg for *T. latifolia* and 1.37 ± 0.06 m^2/kg for *T. angustifolia*. The fraction of end-of-season biomass in each of the first three 10 cm layers was 6.7 ± 0.2% for *latifolia* and 7.0 ± 0.3% for *angustifolia*. In other words, most of the biomass of emergent plants were above the 30 cm plane.

5. A biomachine growth model

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. Uptake by biota, including bacteria, algae, and duckweed, as well as cattails, forms an initial removal mechanism. 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. 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).

5.1. *Model structure*

In this highly aggregated, simple model, is assumed that growth occurs in response to nutrient availability, which is considered to be phosphorus. The inhibited form of Malthus' law, long used to describe population growth (Bailey and Ollis, 1986), is selected as the growth model:

$$\frac{N}{t} = \frac{m}{N_{max}}(L - N)N \tag{3}$$

where C = water phase nutrient concentration (gP/m^3); L = upper limit of biomass that can exist at a given C (g/m^2); m = biomass loss rate constant ($year^{-1}$); N = total biomass (g/m^2); N_{max} = maximum biomass that can exist per unit area (g/m^2); t = time (year).

In this equation, and those that follow, biomass (g) is a dry weight. This growth equation is spatially variable, because the water phase concentration that drives growth changes with spatial position along the gradient. The upper limit of possible biomass is a function of this concentration. Based upon calibration information, the form of this dependence is

$$L = N_{max}\left(\frac{(C - C_0)}{(C - C_0) + s}\right) \quad (4)$$

where C_0 = ecosystem extinction threshold concentration (gP/m^3); s = biomass half saturation concentration (gP/m^3).

Several authors have speculated that L should contain a Monod factor for the limiting nutrient (Tilman, 1982; Thomann and Mueller, 1987). The limiting biomass density is constrained by space for stems, and by available light. The water phase concentrations along the flow direction are known from field sampling in each of the 30 operational years. Consequently, Eq. (4) was fit to data, with N_{max} = 9000 g/m^2; C_0 = 0.025 gP/m^3; and s = 1.0 gP/m^3. For purposes of evaluation, it should be noted that the maximum aboveground macrophytes crop was found to be only 2500 g/m^2, with the balance as roots, rhizomes, algae, duckweed, bacteria and fungi. Details of the calibration may be found in Kadlec (1997).

At any particular time and location, the nutrient content of the vegetation may be calculated as the product of a measured tissue concentration and the corresponding biomass.

$$M = x_N N \quad (5)$$

where M = biomass phosphorus content (gP/m^2); x_N = P concentration in biomass (gP/g).

This model may also be used to estimate the amount of nutrient that is sequestered as newly created soils and sediments. These are the residuals from litter decomposition. The required parameters are the rate of biomass turnover and the fraction of the biomass that is refractory to decomposition:

$$S = k_N N = [x_N m(1 - \beta)]N \quad (6)$$

where m = loss rate constant for biomass ($year^{-1}$); k_N = burial rate constant (gP/g year); S = net phosphorus removal rate (gP/m^2 year); β = fraction of P returned to water by leaching and decomposition.

The assumption is that the local accretion flux is proportional to the lumped biomass at that given location in the wetland. This approach has been described for rivers, in which the attached plant biomass is assumed to be the primary determinant of phosphorus uptake (Thomann and Mueller, 1987).

Collectively, Eqs. (3)–(6) are termed a "biomachine" model, because the wetland vegetation and other biota serve as a mechanism to take up nutrients, and deposit a small fraction as new soils and sediments. These partial differential equations are solvable by numerical methods, using spreadsheets.

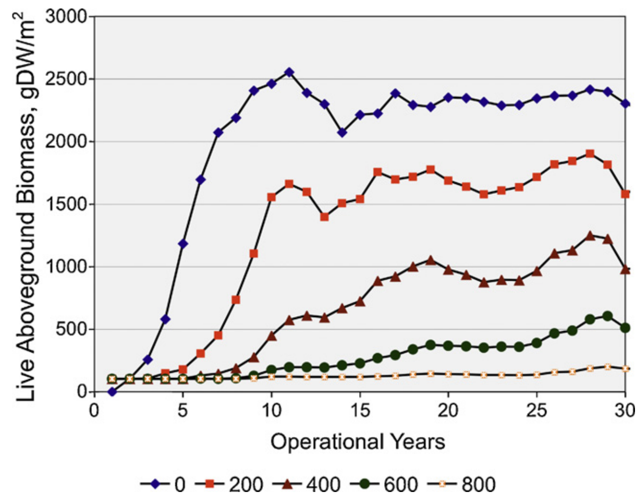

Fig. 15 – Aboveground live plus standing dead biomass as a function of time at different distances from discharge (m). These are model results from the biomachine model.

Here this model is used to provide a means of interpreting the biomass in the irrigation zone as a function of time and spatial position. However, the implications for the storage of nutrients in the new soils and sediments is explored further in Kadlec (2009-b).

5.2. *Model results*

The water phase concentrations drive the model, creating increases in biomass over the initial, pre-project condition. In the calibration of the model, the biomass response at the discharge line was used as a primary objective. The field data response is shown in Fig. 12, together with the model representation. The model forecast line is not smooth, because of the year-to-year variability of the water concentration gradients that force it. It is seen that only the general grow-in trend is represented, and not the oscillations that result from unknown and unpredictable factors. However, the initial increases in biomass, over the first 10 years, are reasonably well represented. It is then possible to infer the responses to fertilization at other distances from the discharge (Fig. 15). Because the zone nearest the discharge line consumes considerable amounts of nutrients (phosphorus), the concentrations down gradient are lower, and produce less of an effect of increased biomass. The appearance of a biomass increase at a downstream location is delayed until the consumption of nutrients in the upstream zone for biomass increase has occurred. At any distance, the biomass grow-in ultimately is complete, and lesser ultimate standing crops result from the lower concentrations in downstream waters. Thus response times are longer, and final biomass values are lower, as distance increase. It is perhaps surprising that the duration of the vegetative grow-in is forecast to be the entire 30 years at distances of 600–800 m.

The cross-plot of model results, with distance as the x-axis and curves for individual years (Fig. 16). The spatial curves show declining biomass with distance. Each year the curves move further downgradient, describing an advancing front of

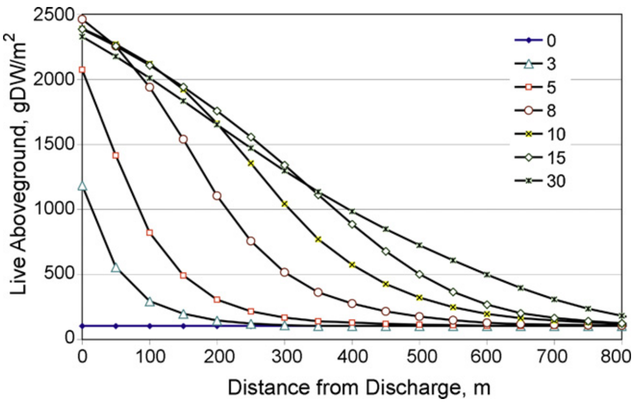

Fig. 16 – Aboveground live plus standing dead biomass as a function of distance from discharge. These are model results from the biomachine model.

biomass. The model does not distinguish among species, but most of that biomass was cattail. This front did not advance without stopping. The 30-year profile is coincident with the long-term limit that would occur if irrigation continued at the same rate, and at the same phosphorus concentration.

It is informative to track the location of fronts as a function of time, because of the intense interest in forecasting nutrient impacts in other systems as well as at Houghton Lake. A particular isopleth of water concentration may be tracked directly from data, and that has been done for smoothed transect phosphorus data for each of the project years, as further described in Kadlec (2009-a). Here, the value of 0.232 gP/m^3 is selected, corresponding to an above ground standing crop (N_a) of 500 g/m^2 in the model (proportioned from Eq. (4)). As indicated by Eq. (3), grow-in is not instantaneous, but requires several years. The results are shown in Fig. 17, where the locations of the concentration front at $C = 0.232$ gP/m^3 are derived from field data, and the locations of the biomass front at $N_a = 500$ g/m^2. Because the biomass response integrates several years of concentration exposure, the calculated values of

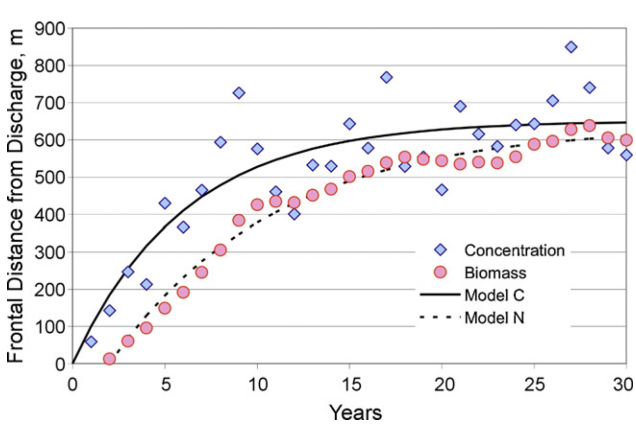

Fig. 17 – Locations of the water phosphorus concentration and biomass fronts as a function of time. The selected concentration value was 0.232 gP/m^3, corresponding to a model value of 500 g/m^2 of aboveground biomass. The lines are regressed from concentration data and biomass model values.

the biomass front display less variability than the concentration front.

The frontal locations follow approximate exponential trends, leading to long-term asymptotes:

$$x = x_0 \left(1 - \exp\left(-\frac{t - t_0}{a}\right)\right) \quad (7)$$

where a = time constant for frontal movement (year); x = location of the front (m); x_∞ = location of the front after completion of movement (m); t = time since discharge began (year); t_0 = time of first response (year).

The regression parameters were $x_\infty = 650$ m, $a = 6$ years, and $t_0 = 0$ year for concentration ($R^2 = 0.73$); and $x_\infty = 640$ m, $a = 9$ years, and $t_0 = 2$ years for above ground biomass ($R^2 = 0.98$). Note that the time constant for response of the calculated biomass front of 9 years matches that determined from aerial photography and Eq. (1).

6. Nutrients and chemicals in vegetation

The phytomass in the Porter Ranch wetland contained a very small standing stock of nitrogen and phosphorus in the pre-project condition. The wastewater additions caused large increases in the tissue concentrations of both nitrogen and phosphorus. When compounded by the large increases in phytomass due to irrigation, there were exceptionally large increases in the standing crops of nitrogen and phosphorus.

Care must be taken in the interpretation of measured tissue nutrient concentrations. Plant growth changes the proportions of stored P in various plant parts as the seasons progress. The decline of above ground tissue nutrient content is a common phenomenon in both treatment and natural wetlands, and results in markedly lower tissue N and P concentrations at the end of the growing season. For instance, measurements have shown concentration decreases of 0.006%DW per day for nitrogen, or a season-long decline of more than 0.5%DW (Kadlec and Wallace, 2008). Other studies have shown the same phenomenon for P concentrations, which can go down over 0.001%DW per day (Bayly and O'Neill, 1972). It therefore apparent that the timing of vegetation samples can greatly affect subsequent calculations of nitrogen and phosphorus storage. Similar phenomena were observed for the Porter Ranch peatland, both before and during irrigation (Wentz, 1975). Therefore, interpretations here are based on end-of-season tissue nutrient concentrations, coinciding with the measurements of the amounts of phytomass in the wetland.

6.1. *Tissue nutrient content*

The nutrient concentrations in various plant parts were measured in pre-project years, and during several project years at control locations (Table 6). Tissue phosphorus was low, averaging 0.086%DW across times and compartments. The P content of cattail leaves was 0.130%DW, which is comparable to the median value of 0.146%DW from other studies of natural wetlands (Kadlec and Wallace, 2008). It is noteworthy that standing dead material is known to have considerably lower P

Table 6 – Biomass nutrient concentrations at the end of the growing season, for controls in 1978–1989, and during the pre-project period 1973–1974. Triplicated samples in each year. Year-to-year S.D. are given when there are multiple measurements. Units are percent dry weight.

	Nitrogen		Phosphorus	
	Mean	S.D.	Mean	S.D.
Sedge-willow				
Sedge	1.15	0.11	0.065	0.018
Grass	0.71		0.061	
Willow	2.46		0.225	
Standing dead	1.10	0.28	0.028	0.004
Litter	1.70	0.25	0.066	0.024
Roots	1.38	0.11	0.040	0.014
Leatherleaf-Bog Birch				
LL	1.61		0.086	
BB	1.81		0.096	
Herbs	1.21		0.078	
Litter	1.22		0.067	
Cattail				
Live	1.47	0.15	0.130	0.035
Litter	2.50	0.14	0.095	0.036
Average	1.52	0.54	0.086	0.051

content, with a median value of 0.051%DW from other studies. Tissue nitrogen averaged 1.52%DW across times and compartments in the pre-project condition and in controls (Table 6). The N content of cattail leaves was 1.5%DW, which is comparable to the median value of 1.8%DW from other studies of natural wetlands (Kadlec and Wallace, 2008). The tissue N and P at backgradient locations, under the influence of longer hydroperiods but no increases in nutrients, were similar to those measured pre-project or in controls (Table 7). During a 4-year period 1995–1998, live cattail tissue P averaged 0.10%DW, and live cattail tissue N averaged 1.07%DW. There were lesser concentrations in standing dead materials.

In general, the tissue nutrient concentrations were much larger in the discharge zone than in the backgradient zone, approximately doubled for nitrogen (2.03%DW vs. 1.07%DW) and tripled for phosphorus (0.33%DW vs. 0.10%DW).

6.2. *Standing crops of nutrients*

The standing crop of a nutrient is the product of the tissue nutrient concentration and the amount of phytomass present. During a 4-year period 1995–1998, the live aboveground biomass in the discharge area was 2.5 times greater than in backgradient areas (Table 7). The amounts of litter are increased more. These effects compound to give extreme differences in the standing crop of nitrogen and phosphorus (Table 7). The standing crops in the discharge zone were 16 gP/m^2 and 126 gN/m^2. There is 20 times as much phosphorus in the live, dead and litter aboveground in the discharge area than in the backgradient area, and 14 times as much nitrogen. The difference is most pronounced for the litter compartment, with 91 times as much phosphorus and 56 times as much nitrogen.

These 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. Gradients in tissue nutrient content were also measured, although only in the early years of the project. For instance, in 1980, the P content of *T. latifolia* decreased from 0.28%DW at the discharge to 0.19%DW at 100 m downgradient, and the N content from 1.7%DW to 1.3%DW. Therefore, gradients in biomass storage of N and P were certain to have existed, although these were measured only sporadically. When these gradients are considered, via the calibrated

Table 7 – Biomass nutrient concentration and content at the end of the growing season in the *T. latifolia* areas during the period 1995–1998. S.D. are high, approximately 50% of the means. Triplicates in each year. Below ground standing crops were not measured, only below ground tissue nutrient content, in 2006.

	Phosphorus		Nitrogen	
	Backgradient	Discharge	Backgradient	Discharge
Biomass (gDW/m^2)				
Live	489	1235		
Standing dead	406	1072		
Litter	87	2351		
Content (%DW)				
Live	0.10	0.33	1.07	2.03
Standing dead	0.05	0.26	0.66	1.55
Litter	0.12	0.36	1.70	3.59
Rhizomes (2006)		0.30		1.33
Roots + soil (2006)	0.11	0.30	2.50	3.19
Crop (g/m^2)				
Live	0.50	3.91	5.2	24.3
Standing dead	0.21	2.80	2.7	17.7
Litter	0.10	9.14	1.5	84.2
Live + standing dead	0.70	6.71	7.8	42.0
Total above ground	0.80	15.9	9.3	126

Table 8 – Mass balances for nitrogen and phosphorus in the 100 ha irrigation area over the entire 30-year period of record.

	Water (pumping season) 1000 m^3/year	Phosphorus (POR) (gP/m^2)	Nitrogen (POR) (gN/m^2)
Inflow	599	56.43	135.0
Rain	341	0.58	13.2
Organic			89.9
Fixation			391.5
Total	940	57.0	629.6
Outflow	470	0.85	1.0
Evapotranspiration	470		
Organic			38.5
Biomass		7.68	27.5
Sorption		2.91	0.2
Burial		45.60	562.4
Total	940	57.0	629.6

biomachine model, the average biomass storages in the entire irrigation area were 7.7 gP/m^2 and 27.5 gN/m^2.

6.3. *Sustainable removal: accretion of nutrients*

Sustainable removal of phosphorus is to burial in the form of new accretions of soils and sediments. This accretion flux may be calculated from the calibrated biomachine model, according to Eq. (6). The calibration presumes that about 20% of the biomass N and P are ultimately accreted at a mean biomass P concentration of 0.25% and a mean biomass N concentration of 3.6%.

The estimated 30-year total amounts of N and P entering and leaving the 100 ha irrigation zone are shown in Table 8. For phosphorus, 45.6 gP/m^2 were accreted as new soils and sediments, or 81% of the 56.4 gP/m^2 loaded into the wetland with wastewater. The corresponding amounts of nitrogen accreted were 562 gN/m^2, which considerably exceeded the amount of dissolved inorganic nitrogen in the inflow, which was only 135 gN/m^2. The balance of the nitrogen needed for growth could have come from mineralization of incoming organic nitrogen (estimated at 90 gN/m^2), or from nitrogen fixation (estimated at 392 gN/m^2). This apparent nitrogen deficit is discussed in more detail in Kadlec (2009-a). 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 (Kadlec, 2009-b).

6.4. *Other chemicals in plant tissues*

The wetland vegetation was checked for a number of other chemical constituents, including halogens, metals and sulfur. Neutron activation analysis was conducted on three occasions, in 1978, 1980 and again in 1991. This methodology checks for the concentrations of 35 elements: Al, As, Ba, Br, Ce, Cs, Cr, Co, Cl, Dy, Eu, Ga, Hf, I, Fe, La, Lu, Mg, Mn, Mo, Nd, K, Rb, Sm, Sc, Se, Na, Sr, Ta, Te, Th, Ti, V, Yb, and Zn. Most of these elements were at low concentrations, near or below detection. Data from 1991 for ten of the more interesting elements are given in Table 9.

The non-metals displaying the greatest differences between discharge and control were chlorine and bromine. Chlorine in live leaves to be much greater than in any other compartment—by one to two orders of magnitude. And the discharge zone shows much higher concentrations of chlorine in every compartment tested. The putative reason is the higher chloride content of the incoming wastewater, compared to background waters (123 g/m^3 vs. 28 g/m^3). The chlorine content of the phytomass is surprisingly high, about 340 g/m^3 in the discharge zone, which is mostly in above ground live leaves. Because of the expected strong gradients across the irrigation zone, the average standing crop of chlorine is about 180 g/m^2. The amount of chloride added to the wetland irrigation area each year with wastewater was about 90 g/m^2 year. The result is that the vegetation stored about 2 years worth of incoming chloride. Because of the cycle of growth, death and decomposition with a small residual, a much smaller amount is sequestered in new sediments. Nonetheless, it appears that the biogeochemical cycle is an important factor in chlorine processing, and that chloride should not be regarded as "conservative" at these concentrations in this environment. Bromine was present at much lower concentrations, but it was also higher in the vegetation compartments near the discharge.

Sodium was one of the prevalent cations in the wastewater. It also was incorporated into tissues, and to a much greater extent in the discharge area compared to the backgradient zone. Of the common heavy metals, aluminum, magnesium and cobalt displayed a consistent pattern with respect to the added water: it is higher in the discharge zone, and aluminum and cobalt were higher in the litter and soil. There were not large trends for any of the elements over the three samplings over 14 years.

Sulfur was present in the incoming water, in the form of sulfate, at concentrations in the lagoon discharge to the wetland of 2.6 mg S/L (SO_4S), and in the rainfall impinging on the wetland (1.6 mg S/L). The resultant areal loading on the 100 ha irrigation area was 1.53 gS/m^2 year, and 73% was removed in the wetland. The measured S content of live above ground cattail was 0.42%DW in the backgradient area, and 0.72%DW in the discharge area. If 20% of the live above ground

Table 9 – Elemental analysis of solids compartments, Porter Ranch wetland, in the backgradient control zone and discharge zone. BDL = below detection limit. Means for triplicate samples taken in 1991.

		Magnesium (mg/kg)	Potassium (%)	Sodium (mg/kg)	Bromine (mg/kg)	Chlorine (mg/kg)
Live leaves	Backgradient	420	2.47	758	24.5	13,300
Live leaves	Discharge	559	3.64	3179	46.3	34,700
Standing dead	Backgradient	213	BDL	151	2.4	195
Standing dead	Discharge	262	0.33	924	10.6	369
Litter	Backgradient	282	BDL	338	43.0	727
Litter	Discharge	443	BDL	1283	49.6	1,105
Root zone material	Backgradient	491	BDL	700	36.6	300
Root zone material	Discharge	695	BDL	1612	50.7	858
		Aluminum (mg/kg)	**Cobalt (μg/kg)**	**Chromium (mg/kg)**	**Manganese (mg/kg)**	**Zinc (mg/kg)**
Live leaves	Backgradient	BDL	BDL	7.2	240	8.6
Live leaves	Discharge	BDL	BDL	6.3	194	9.1
Standing dead	Backgradient	72	178	6.9	124	6.6
Standing dead	Discharge	87	259	18.6	54	BDL
Litter	Backgradient	1,008	988	16.0	183	21.0
Litter	Discharge	2,508	2260	10.1	196	33.7
Root zone material	Backgradient	7,765	1083	18.8	32	81.8
Root zone material	Discharge	10,156	2850	14.8	79	67.2

biomass accreted each year, the loss of sulfate sulfur would be accounted.

7. Discussion

7.1. Ecological consequences

Pre-project researchers did not anticipate the large changes in the species composition of the vegetation, in which *Typha*-dominant communities replaced sedge-shrub dominant communities. Consequently, studies focused on relative growth rates and tissue nutrient concentrations, and not on community shift dynamics (Wentz, 1975; Richardson et al., 1978). However, those pre-project studies were essential to defining the original ecological condition of the wetland. Further, the original vegetation research experimental designs contemplated study of vegetation associations only within about 200 m of the discharge line, which proved to be considerably less than the region of impact, and did not include the fringe ecotones that developed for the ultimate grow-in and expansion of the impacted zone.

The area immediately near the discharge became a dense stand of *Typha*, with low species diversity. Over 30 species were routinely found on plots in the original cover types in the irrigation area, and in areas of similar cover but remote from the discharge after the project was in operation. Wentz (1975) tabulated 69 species occurring at relative abundance greater than one percent, over the entire peatland, most of which occurred in or near the irrigation area. By 1992, only eight species were found on the plots in the area near the discharge. Interestingly, these plots did not include *Typha angustifolia*, which is becoming a large fraction of the cover in the irrigation area.

A number of other reports of cattail invasion due to wastewater addition have been published (Cooke et al., 1990; Kadlec and Bevis, 1990; Lowe and Keenan, 1997; Rutchey et al., 2008). These agree that the increased hydroperiods and/or higher nutrient availability give a competitive advantage to *Typha*.

Intrageneric competition is known to occur between *T. latifolia* and *T. angustifolia*. *T. latifolia* is competitively superior in water shallower than 15 cm, with *T. angustifolia* dominating in deeper water, and *T. angustifolia* has a higher tolerance for salinity (Shih and Finkelstein, 2008). It has been observed that within the region, widespread invasion of *T. angustifolia* is typically delayed with respect to its first presence, by two or three decades (Shih and Finkelstein, 2008).

Of special interest are the fringe zones of the impacted area. These are typically a few hundred meters in width, and contain both the original species and a visually obvious proportion of *Typha*. The numbers of species found on plots in these ecotones, in abundances greater than 1% on at least one plot, were: 17 in the sedge-cattail fringe; 19 in the bog birch-cattail fringe, and 28 in the leatherleaf-cattail fringe. In comparison, control plots had 36 species on sedge-bog birch plots, and 27 species on leatherleaf-bog birch plots. Thus the heavily impacted area had much reduced numbers of species (8), but the fringe areas maintained most of their species despite cattail invasion (17–28 out of 27–36).

7.2. Floating mats

Floating mats must be almost entirely organic in order to be buoyant enough to float. They derive their buoyancy from gas spaces in rhizomes (Hogg and Wein, 1987, 1988; Krusi and Wein, 1988), and also from gases generated by decomposition processes.

There are several natural mat formation mechanisms:

(1) The delamination and floating of unvegetated organic substrates from deeper sediment. Germination of plants occurs after emergence. This is a peat float-up process.

(2) The rhizomes of aquatic plants colonise the water surface from a nucleus of aquatic vegetation that is either unattached or expanding from the shore. This is the grow-over process.
(3) Units of rooted vegetation and substrate split simultaneously from the bed, and float to the wetland surface. This is a mat floating process.
(4) Upward root retreat, with accompanying detachment from underlying soils.
(5) Dissolution or destabilization of the rooting soil via chemical processes. Notably, peat soils are acidic, and lose structure upon exposure to alkaline waters.

The first three of these were identified by Clark and Reddy (1998). The first mechanism was not operative at the Porter Ranch, because there were no areas of unvegetated peat in the wetland at any time. Mechanism (2) did occur to some extent. Open water zones near the discharge were covered by a thick (10–15 cm) layer of *Lemna* spp., and that mat fostered *Typha* seed germination and seedling development. Seedling densities up to 100 per square meter were observed. Because there were only limited areas of open water at any time in the irrigation area, it is probable that mechanism (3) was a principal factor in the cattail mat formation. Another possible cause was the retreat of the root zone to a smaller biomass located high in the soil profile, compared to prior conditions (mechanism 4). The original sedge root mat penetrated approximately 50 cm, whereas the cattail rooted to only about half that thickness. Loss of the deep portion of the root mat could have contributed to delamination. Finally, chemical phenomena may have contributed to a new soil structure. When the sedge peats were exposed to basic (alkaline) water in the laboratory, they changed from granular soils to a loose gelatinous ooze. The added wastewater was slightly alkaline (pH ≈ 8.0) compared to the original peat (pH ≈ 6.0), and could have "titrated" the soils to an unstable condition.

Other wastewater-impacted wetlands have experienced mat development. For instance, the natural wetland at Kinross, Michigan wetlands also developed a near-monoculture of cattails on a pre-existing peatland. Over the course of time, this *Typha* community became a floating mat (Kadlec and Bevis, 1990). This physical effect, coupled with possible partial peat dissolution into the less acidic added wastewater, led to a 50 cm soil-free water zone topped by the floating mat. The mats are closely woven beds of roots, rhizomes and sediments.

The development of the mat has important implications for treatment. The impacted zone of the pre-mat wetland contained a biologically active water column community dominated by algae and duckweed, together with the associated bacteria and fungi. These cycled N and P, and contributed to the development of acting residuals. The mat environment is extremely hostile to algae, because of light limitation for under-mat flows. Thus incoming algal materials were likely killed and filtered in an extremely small zone near the discharge. Without the presence of the duckweed-algae component, it is possible that the treatment effectiveness of the wetland has been impaired in the mat region. That region is still the minority of the irrigation zone, however. There are also compensating factors that may improve treatment in the mat configuration. The water flow is more intimately connected to the root zone, thus improving the ability of the macrophytes to access nutrients. The water is exposed to soils on both its upper and lower surfaces, increasing the opportunity for soil processes to provide treatment. There is also undoubtedly a microbial community under the mat that contributes to treatment. Thus, the net effect of the mat development on treatment is largely unknown at this time. Parenthetically, it is noted that constructed mat wetlands have found favor for wastewater treatment elsewhere (Hiley, 1990; Smith and Kalin, 2002; Richter et al., 2003; Curt et al., 2005), although not necessarily for phosphorus removal.

7.3. *The role of plants in nutrient reduction*

The mechanisms that remove P in marsh ecosystems are in three categories: (1) sorption on antecedent substrates, (2) storage in an increased standing crop of biota, and (3) the formation and accretion of new sediments and soils (Kadlec, 1997). The first two of these processes are saturable, meaning they have finite capacity and therefore cannot contribute to long-term, sustainable P removal. Nitrogen removal involves those same mechanisms, but also undergoes losses due to denitrification and possible gains due to fixation. Biomass increases form the second part of ecosystem luxury uptake. Macrophytes, algae and microbes all utilize phosphorus as an essential nutrient, and contain P in their tissues. New sources of P typically foster increased standing stocks in live above and below ground biomass, which in turn create larger stocks of standing dead and litter material. Net P removal by this mechanism ceases when the total new biomass pool has reached a new larger, sustained size. At Houghton Lake, 14% of the added phosphorus and 20% of the added nitrogen were sequestered in the new, larger standing crops over the 30-year POR (Table 8). Only minor amounts were stored due to sorption, 5% for phosphorus and essentially zero for nitrogen. Mechanism three was dominant for phosphorus, and also the major storage for nitrogen. Therefore, in the Houghton Lake wetland, the removals of nitrogen and phosphorus were almost totally controlled by processes involving the vegetation and associated biota. The grow-in of the new large standing crop was not instantaneous, but took place over about 9 years near the discharge, and was still occurring after 30 years at the downstream edges of the impacted area.

This wetland was, in the parlance of treatment wetland literature, lightly loaded with nutrients. For example, the nitrogen loading to the 100 ha irrigation area was 4.5 gN/m^2 year. This was far below the agronomic criterion of 120 gN/m^2 year suggested by Kadlec and Wallace (2008). The phosphorus loading of 1.9 gP/m^2 year was also low on a relative basis, at about the 30th percentile of other treatment wetlands. These low loadings took this project out of the realm of treatment wetlands applied to secondary activated sludge or lagoon effluents, and invalidate the statements made in several literature sources. For instance, Crites et al. (2006) state that: "Nitrogen removal in constructed wetlands is accomplished by nitrification and denitrification, Plant uptake accounts for only 10% of the nitrogen removal." At Houghton Lake, virtually 100% of the nitrogen removal was due to plant growth and burial. USEPA (2000) identify two "common misconceptions": (1) that "Constructed wetlands can remove significant amounts of nitrogen," and (2) that "Constructed wetlands can remove significant amounts of phosphorus." Comment (1) was predicated on the concept that nitrification–denitrification is the primary mechanism. The irrigation area at Houghton Lake removed 95%. Comment (2) was predicated on the concept that plant uptake is minor, and negated during senescence. The irrigation area at Houghton Lake removed 94%.

It is probable that little or no nitrification occurred in the irrigation zone, because of the anoxic conditions (Kadlec, 2009-a, 2009-b). However, the small amount of incoming nitrate (about 1.6 mg N/L) would have been exposed to near-optimal conditions, of adequate carbon and low redox potential (Kadlec, 2009-a, 2009-b). However, the amount of nitrogen required for plant growth is defined by the measured biomass cycle, and exceeded the amount of incoming ammonia (Table 8). Therefore, another source of nitrogen, i.e., fixation must have supplied the difference.

7.4. Litter storage

Large amounts of added nutrients were stored in an increasingly large litter compartment. Litter decomposition reported for *Typha latifolia* in other studies is higher than that determined in this study. Chimney and Pietro (2006) report a mean of 2.63 $year^{-1}$, and a median of 1.13 $year^{-1}$, for 14 studies of decomposition of *T. latifolia*. This literature does not use an initial weight loss. When the *T. latifolia* data in this study is analyzed with $W_0 = 100\%$, the decay coefficients are 0.55 $year^{-1}$ near the discharge, and 0.73 $year^{-1}$ for the backgradient location. At this cold-climate location, the litter layer was either frozen or at near 0 °C temperature for half the year. Therefore the average warm-weather disappearance rate coefficients are roughly double the values determined on an annual basis. This would place the results of this study near the median of other such studies.

Not all of the leaf and wood litter are expected to decay and disappear. There are certain to be residuals that are resistant to decay. Starches and sugars are removed, leaving lignin and other refractory materials. These comprise the new sediment and soil accretions that occur in peatlands of long hydroperiod. Eq. (2) does not describe or allow for this undecomposed residual, but it is probable that the period of data acquisition was not long enough to approach completion of decay processes. When such residuals accrete at the top of an underwater soil surface, they become part of the rooting medium for the macrophytes. However, in a floating mat situation, the water and nutrients are below the top of the sediment/soil surface, and roots develop in locations below the horizon at which litter accumulations occur. Speculatively, the litter layer on top of the floating mat remains loose, and unconsolidated by new root growth. This could account for the increasing time trend observed for the standing crop of litter in the mat zone (Fig. 13).

7.5. Containment of impacts

The result of new P loadings was the creation of zonation within the receiving wetland, with stronger effects near the point of discharge, diminishing in the direction of water flow as P is stripped from the water. In general terms, the zone nearest the new discharge may undergo species alteration; zones further away may retain their species under nutrient enrichment, and at long distances the background ecosystem will continue to prevail (Lowe and Keenan, 1997). There is a zone in which vegetative alterations are contained, here termed the zone of total containment (ZTC). This zone will grow until removal mechanisms balance additions, reaching a final size termed the *ultimate zone of total containment* (UZTC). The ZTC will be imbedded in the antecedent marsh conditions, which may be either impacted or unimpacted; in the case of Houghton Lake, the wetland was unimpacted in its original condition.

The impacted zone may, in some circumstances, exceed the confines of the original marsh boundaries (Lowe and Keenan, 1997). For instance, the Kinross, MI project discharged approximately 900,000 m^3/year of wastewaters with a P content of about 8 gP/m^3 into a natural peatland for several decades (Kadlec and Bevis, 1990). The resultant annual phosphorus loading was about triple that at the Houghton Lake system. The impacted area grew to approximately 100 ha over 30 years, but that area did not contain the impacts, but rather was defined by a downstream boundary, in the form of a freeway constructed across the wetland.

It is also important to note that the scale of impacts is determined by the biology and ecology of the wetland, and not by the presence of real or imagined boundaries. The 83 ha visually impacted area resulted from an average annual loading of 1.9 metric tons of phosphorus. Therefore, the irrigation zone contained the ecological effects for an average areal loading of 2.25 gP/m^2 year, and reduced concentrations to background levels for three decades. The 83 ha were an approximation of the ultimate zone of total containment (UZTC). If a smaller treatment zone is arbitrarily defined, then that zone will not provide total containment. For instance, Richardson and Marshall (1986) arbitrarily chose a 19.5 ha "test area," and used project data to conclude that it had become "saturated" in 1980, because a portion of the applied phosphorus was moving further downstream. Based on soil sorption alone, Richardson and Marshall (1986) concluded that this wetland could store 1.0–1.5 gP/m^2, and that "vast acreages" would be needed for storage of phosphorus at these "high levels of phosphorus." Richardson and Marshall (1986) did not consider that portion of the project data that reported the huge new standing crops of new plants (cattails), which have here been shown to contain 16 gP/m^2.

Although the temptation is very great, it is not appropriate to conclude that the average annual Houghton Lake ZTC loading of 2.25 gP/m^2 is a universal number, or even a locally generalizable number. It is at best a benchmark for a specific set of conditions. There were strong gradients of biomass and concentrations, meaning that removals were about 5 gP/m^2 year near the discharge line, and only about 1 gP/m^2 year near the outlet edge of the irrigation zone. Given the Monod characteristic of the presumed biomass model, removals were near-zero order near the inlet, and near-first order near the edge. That in turn implies that removal does not scale linearly with concentration, and therefore average areal loading is an inadequate scaling variable for either water quality or for ecological impacts.

7.6. Moving fronts

The size of the ZTC grew during the POR, with the entire 30-year POR needed to reach the final size. This study provides strong indications that a wetland does not incur boundless alteration due to relatively constant nutrient additions. Rather, an impacted zone develops to a size sufficient to strip the added nutrients from the water. The long-term sustainable mechanism for phosphorus removal is the creation of new sediments and soils, which accrete in the impacted zone. Within that zone, there are strong gradients in concentrations and the resulting biomass and species composition. However, this study also establishes that the periods of time over which alteration takes place are quite long, and are exacerbated by the 6-month growing season in north temperate climate zones. The 9-year time constant for impact development has important implications for interpretation of information from other wetlands under similar circumstances. No short-term study, i.e., a year or two, can hope to quantify biological effects that take many years to manifest.

There was an advance of the P front during the transition to steady state in the Porter Ranch peatland under the imposed loading. There has been speculation by some that the movement of such new phosphorus "fronts" will continue indefinitely, and reach to ever increasing distances with the passage of time (Gaiser et al., 2005). However, mass balance considerations contradict this concept, because if there is removal, then there must exist an area within which that removal balances the added inputs. The new P front advances only to the point at which the sustainable P removal mechanisms balance the new P additions. The data in this study support this concept. However, the transition to the final steady state occupied a fairly long period of time (years), and is accompanied by alteration of the ecosystem within the zone upstream of the front. Richardson and Qian (1999) recognized this, and stated: "If the wetland is of sufficient acreage . . . then a high percentage of the P will be retained within the wetland and no moving P front in the water column will exist."

7.7. Stability of the impact zone

The stability of a ZTC after establishment is conditional on long hydroperiods. Continual inundation of the marsh creates the conditions that promote accretion, and hence sustainable P removal. Oxidation of the organic matrix that contains the unavailable organic phosphorus can reverse this storage. The oxidation can be slow, due to exposure of the upper soil horizon to air; or rapid, in the event of a peat fire. In either case, stored P is mineralized, and made available for easy export upon reflooding.

The stability of the ZTC is also contingent on containment of particulate solids movement. There is a strong likelihood that high flow events, bioturbation and other natural phenomena are capable of displacing some portion of the flocculent material that builds in a nutrient-rich marsh. Further, it is possible that high-turnover processes, such as microbial uptake and release, could contribute to this nutrient "spiraling" (Gaiser et al., 2005). It is probable that floc movement is much slower than water movement, because it occurs by detachment and re-settling processes that have limited range for any given event. The terminal beaver impoundments at Houghton Lake could have played a role in reducing export of these movable solids. In any case, dense marsh vegetation is known to be effective in stopping transport of particulates.

A key question remains unanswered after this project has operated for the prolonged period of record: if the wastewater loading is terminated, what will be the response of the impacted zones of the ecosystem? It is intuitively clear that the impacted area will lose some of its nutrients if exposed to clean water flushing, from rainfall or other sources. The nutrients contained in the litter are to some extent leachable, and therefore capable of remobilization. Indeed, in this project, lab research showed that up to 700 mgP/kg litter could be rinsed from cattail litter, but less than 100 mgP/kg of peat could be rinsed out. These values were the extrapolated limits of indefinite rinsing. These amounts are only a tiny fraction of the total storages of phosphorus, presumably stored by sorption processes.

The nutrients stored in expanded biomass would be returned primarily to the water column during a biomass "spin down" after cessation of fertilization. The biomachine model might be used to roughly estimate the speed and extent of this process, but there are no calibration data. The removal of the nutrient source would be immediately seen in the first seasonal growth event, if the nutrients were solely withdrawn from that source. However, some of the soil-bound nutrients are likely to be accessible by the plants, and that would indicate a biomass response spanning possibly spanning several growing seasons. It is not known what fractions of the stored nutrients are accessible by macrophyte "mining."

Given that there would be an eventual collapse of the biomass due to removal of fertilization, there is yet more uncertainty about the extent and speed of possible species composition changes. The cattails may or may not disappear in the face of lesser nutrients.

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