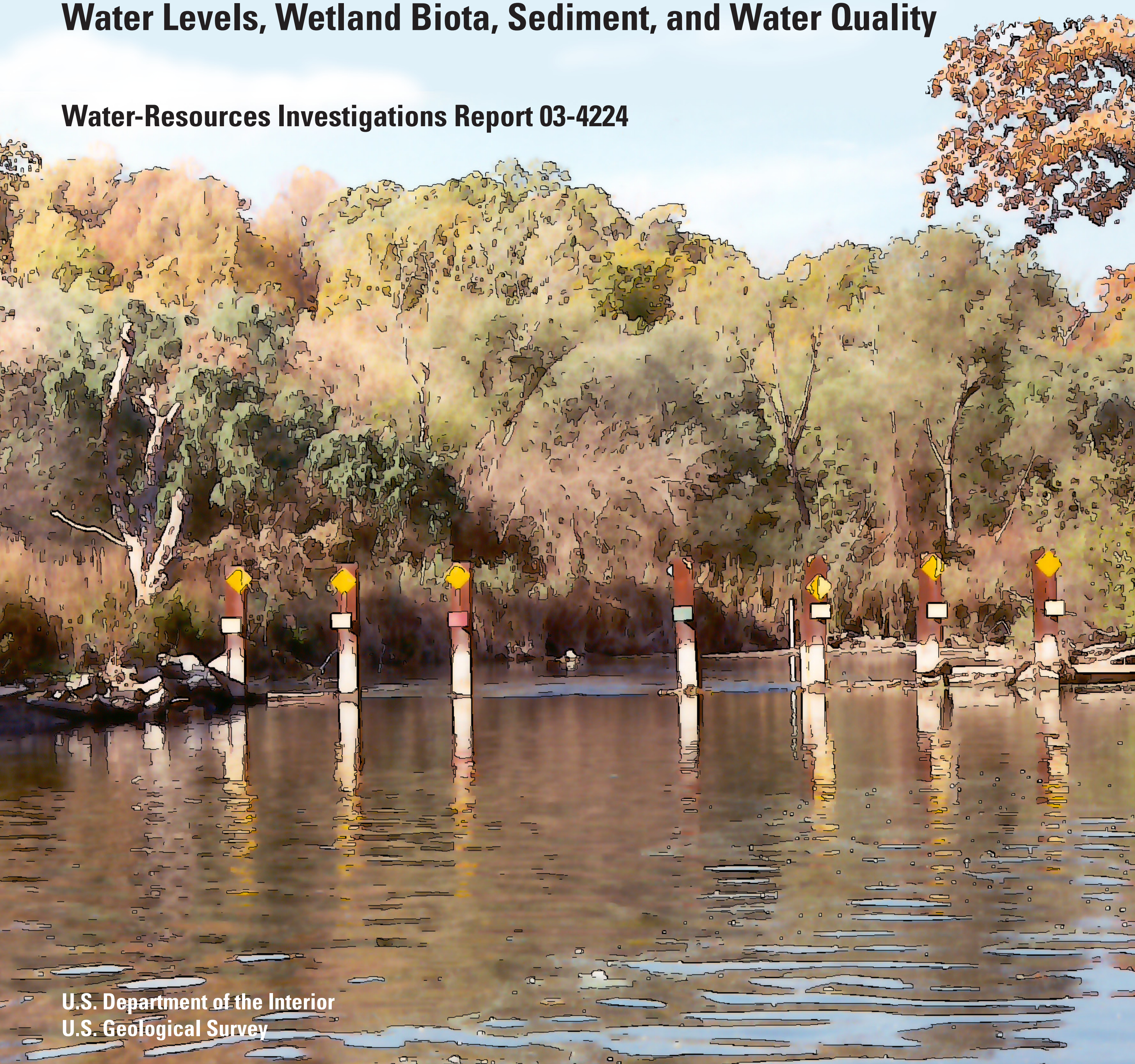
In cooperation with the
MONROE COUNTY DEPARTMENT OF HEALTH

Effects of Flow Modification on a Cattail Wetland at the Mouth of Irondequoit Creek near Rochester, New York

Water Levels, Wetland Biota, Sediment, and Water Quality

Water-Resources Investigations Report 03-4224



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Effects of Flow Modification on a Cattail Wetland at the Mouth of Irondequoit Creek near Rochester, New York

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By WILLIAM F. COON

In cooperation with the Monroe County Department of Health

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U.S. Department of the Interior
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Conversion Factors and Datum

INCH-POUND TO INTERNATIONAL SYSTEM (SI) UNITS

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	4,047	square meter (m^2)
square foot (ft^2)	0.09290	square meter (m^2)
square mile (mi^2)	2.590	square kilometer (km^2)
	Volume	
cubic foot (ft^3)	0.02832	cubic meter (m^3)
acre-foot (acre-ft)	1,233	cubic meter (m^3)
	Flow rate	
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
	Mass	
ounce, avoirdupois (oz)	28.35	gram (g)
pound, avoirdupois (lb)	0.4536	kilogram (kg)
ton, short (2,000 lb)	0.9072	megagram (Mg)

INTERNATIONAL SYSTEM (SI) TO INCH-POUND UNITS

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
	Area	
hectare (ha)	2.471	acre
square meter (m^2)	10.76	square foot (ft^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
cubic meter (m^3)	35.31	cubic foot (ft^3)
cubic meter (m^3)	0.0008107	acre-foot (acre-ft)
	Flow rate	
cubic meter per second (m^3/s)	35.31	cubic foot per second
	Mass	
gram (g)	0.03527	ounce, avoirdupois (oz)
kilogram (kg)	2.205	pound, avoirdupois (lb)
gram per square meter (g/m^2)	.0002049	pound, per square foot (lb/ft^2)
megagram (Mg)	1.102	ton, short (2,000 lb)

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25°C).

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:
°F = (1.8 x °C) + 32

Time: h, hour; min, minute; s, second

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L).

Vertical coordinate information is referenced to the *National Geodetic Vertical Datum of 1929 (NGVD of 29).*

Horizontal coordinate information is referenced to the *North American Datum of 1927 (NAD 27).*

Effects of Flow Modification on a Cattail Wetland at the Mouth of Irondequoit Creek near Rochester, New York—Water Levels, Wetland Biota, Sediment, and Water Quality

by William F. Coon

ABSTRACT

An 11-year (1990-2001) study of the Ellison Park wetland, a 423-acre, predominantly cattail (*Typha glauca*) wetland at the mouth of Irondequoit Creek, was conducted to document the effects that flow modifications, including installation of a flow-control structure (FCS) in 1997 and increased diversion of stormflows to the backwater areas of the wetland, would have on the wetland's ability to decrease chemical loads transported by Irondequoit Creek into Irondequoit Bay on Lake Ontario. The FCS was designed to raise the water-surface elevation and thereby increase the dispersal and detention of stormflows in the upstream half of the wetland; this was expected to promote sedimentation and microbial utilization of nutrients, and thereby decrease the loads of certain constituents, primarily phosphorus, that would otherwise be carried into Irondequoit Bay. An ecological monitoring program was established to document changes in the wetland's water levels, biota, sedimentation rates, and chemical quality of water and sediment that might be attributable to the flow modifications.

Water-level increases during storms were mostly confined to the wetland area, within about 5,000 ft upstream from the FCS. Backwater at a point of local concern, about 13,000 ft upstream, was due to local debris jams or constriction of flow by bridges and was not attributable to the FCS.

Plant surveys documented species richness, concentrations of nutrients and metals in cattail tissues, and cattail productivity. Results indicated that observed differences among survey periods and between the areas upstream and downstream from the FCS were due to seasonal changes in water levels—either during the current year or at the end of the previous year's growing season—that reflected the water-surface elevation of Lake Ontario, rather than water-level control by the FCS. Results showed no adverse effects from the naturally high water levels that prevail annually during the spring and summer in the wetland, nor from the short-duration increases in water levels that result from FCS operation.

Fish surveys documented the use of the wetland by 44 species, of which 25 to 29 species were found in any given year. Community composition was relatively consistent during the study, but seasonal and year-to-year variations in dominant resident and nonresident species were noted, and probably reflected natural or regional population patterns in Lake Ontario and Irondequoit Bay. The FCS allowed fish passage at all water levels and had no discernible adverse effect on the fish community.

Bird surveys documented the use of the wetland by more than 90 species for breeding, feeding, and migration. Ground-nesting birds were unaffected by the FCS. Seasonally high water levels, rather than short-duration increases caused by the FCS, might have caused the scarcity or absence of certain wetland species by limiting the extent of breeding habitat for some species and the exposure of mud flats that attracted other species. Some noticeably scarce or absent species also were rare or absent elsewhere along the south-central shore of Lake Ontario.

Benthic-macroinvertebrate studies were of minimal use for evaluating the effect of the FCS because no surveys were conducted after FCS installation. The precontrol results allowed assessment of the ecological quality of the wetland on the basis of biotic indices, and generally indicated moderately to severely impaired conditions. Differences between the macroinvertebrate communities in the southern part of the wetland and those in the northern part were attributed to habitat differences, such as substrate composition, water depth, and density of submerged aquatic vegetation.

Sedimentation rates in the areas upstream and downstream from the FCS increased after the flow modifications, more in the area upstream from the FCS than in the downstream area. The concurrent downstream increase and the dynamic patterns of deposition and scour indicated that although the FCS and the other flow modifications undoubtedly were major factors in the postcontrol upstream increase in sedimentation rates, other factors, such as the magnitude, frequency, and the timing (season) of peak flows, might also have contributed.

Periodic analyses of sediment samples from three long-term depositional sites in the wetland documented the concentrations of major and trace elements, polycyclic aromatic hydrocarbons, and organochlorine and organophosphate compounds. The concentrations of most constituents showed no substantial fluctuation or consistent upward or downward trend

during the years sampled, nor did they identify any change after FCS installation. Comparison of the measured concentrations with sediment-quality guidelines that are used to assess the ecological quality of substrate environments indicated that the wetland was moderately to severely impaired—an assessment consistent with the benthic-macroinvertebrate biotic indices.

During the precontrol period (1990-96), the wetland was a sink for particulate constituents (removal efficiencies for total phosphorus and total suspended solids were 28 and 47 percent, respectively), but had little effect on conservative constituents (chloride and sulfate). The wetland was a source of orthophosphate and ammonia (removal efficiencies were -38 and -84 percent, respectively).

During the postcontrol period (1997-2001), the wetland continued to be a sink for particulate constituents (removal efficiencies for total phosphorus and total suspended solids were 45 and 52 percent, respectively); the exportation of orthophosphate by the wetland decreased (by 7 percent), whereas that of ammonia increased (by about 70 percent). The outflow loads of orthophosphate and ammonia represented about 15 and 2.3 percent of total phosphorus and total nitrogen loads, respectively. Changes in the loads of conservative constituents were negligible, and the overall removal efficiencies for other constituents during the precontrol period differed from those of the postcontrol period by no more than 5.4 percent.

Statistical analyses of monthly inflow and outflow loads indicated significant differences between inflow and outflow loads of most constituents during the pre- and postcontrol periods. Load data were adjusted to remove the effects of dissimilar hydrologic conditions that prevailed during the pre- and postcontrol periods, and to isolate the water-quality-improvement effect that could be attributed solely to the FCS. Results indicated that the FCS contributed significantly to the decrease in total phosphorus loads, and slightly to a decrease in ammonia-plus-organic nitrogen loads, but had little or no significant effect on loads of other constituents.

Introduction

Storm runoff from urban areas causes local flooding and carries large amounts of contaminants and sediment to downstream receiving water bodies (O'Brien and Gere, 1983; U.S. Environmental Protection Agency, 1983). Runoff-management practices such as stormflow-detention basins and erosion-control measures are routinely implemented to mitigate these adverse effects (Natural Resources Conservation Service, 1986). One practice that has been considered a cost-effective means of decreasing chemical loads in urban runoff is stormwater detention within wetlands, which have the potential to attenuate peak discharges, detain stormflows, and improve stormflow quality by facilitating the retention or removal of sediment and nutrients (Kadlec and Knight, 1996; Coon and others, 2000; U.S. Environmental Protection Agency, 1993; Strecker and others, 1992). This latter function can be achieved through several processes, including sedimentation (settling), adsorption, chemical precipitation, filtration, biochemical interactions and microbially mediated transformations, nutrient assimilation, volatilization and aerosol formation, infiltration (Strecker and others, 1992), ion exchange, biodegradation (U.S. Environmental Protection Agency, 1993), and long-term storage of chemicals through accretion and burial (Mitsch and Gosselink, 1986). These processes can cause a wetland to act as either a temporary or permanent sink for certain constituents. Conversely, other wetland processes, such as leaching, decomposition, dissolution, diffusion from sediments, and, again, microbially mediated transformations, can cause a wetland to act as a source of certain constituents and to export them to receiving waters downstream (Kadlec and Knight, 1996; Strecker and others, 1992; Mitsch and Gosselink, 1986; U.S. Environmental Protection Agency, 1993).

The water-quality-improvement function of natural and constructed wetlands has been studied since the 1960's; wetlands are receiving wide use as a cost-effective and environmentally beneficial means of mitigating the deleterious effects of contaminated runoff and wastewater on surface-water and ground-water systems (Kadlec and Knight, 1996; Strecker and others, 1992; U.S. Environmental Protection Agency, 1993; Livingston, 1989; Gadbois, 1989). Of the processes listed above, microbial activity—the assimilation and transformation of nutrients by microbes—has been identified by many researchers as the principal mechanism by which water quality can be improved through detention in a wetland (Mitsch and Gosselink, 1986; Johnston, 1991; Hickok and others, 1977). The effectiveness of this mechanism might depend on macrophyte density—that is, the amount of surface area to which bacteria, fungi, and epiphytic algae can adhere (Kadlec and Knight, 1996). Physical processes, such as adsorption and sedimentation, also can play major roles in water-quality improvement (Johnston, 1991).

The processes of water-quality improvement in a wetland are interrelated, and the relative importance of any one mechanism can vary from wetland to wetland. The broad range in the effectiveness of wetlands in treating stormwater results from (1) their diversity in terms of vegetation, depth and duration of inundation, climatic factors, soils, and hydraulic characteristics; and (2) differences in water sources (surface water, ground water, or atmospheric deposition) and water chemistry. The net effect of a wetland on water quality depends on the balance between chemical uptake and retention on the one hand, and chemical release and export on the other. Chemical retention or export can be affected by many factors, including:

-size and water-storage capacity of the wetland;
-rate and velocity of flow;
-detention time of stormwater;
-pattern and degree of flow dispersal or mixing (circulation);
-presence or absence of pools for sediment accumulation;
-frequency, duration, and depth of inundation;
-types and density of vegetation;

-water pH, temperature, and dissolved-oxygen concentration;
-turbidity or amount of light penetration;
-season;
-types of maintenance practices used (removal of sediment or vegetation); and
-rates of chemical exchange between water and sediment (Livingston, 1989; Strecker and others, 1992; Kadlec and Knight, 1996; Adamus and Stockwell, 1983).

A wetland's structure and ability to improve water quality depend largely on its hydrologic characteristics, which, in turn (1) determine or modify the composition, species richness, and distribution of wetland flora; (2) affect the level of productivity through volume and circulation of flow and through duration and frequency of inundation; and (3) affect (a) rates of sedimentation, resuspension, and erosion; (b) rates of organic-matter accumulation and export; and (c) the input, output, transformation, and cycling of nutrients and other constituents (Mitsch and Gosselink, 1986).

The use of natural wetlands for treatment of stormwater is less common than use of constructed wetlands because State and Federal regulations restrict the intentional use of natural wetlands for stormwater treatment (United States, 1977; New York State, 1980). The unintentional use of a wetland for this purpose can be unavoidable, however, depending on its location within its basin and its proximity to urbanized areas. When this occurs, the hydrologic, biological, and chemical characteristics of the wetland will be affected by the increased inflows and chemical loads (U.S. Environmental Protection Agency, 1993), and a new equilibrium condition will become established. One such natural wetland that reflects several decades of increasing upstream development is the Ellison Park wetland, at the mouth of Irondequoit Creek on Irondequoit Bay, near Rochester, N.Y. (fig. 1).

Irondequoit Creek Basin

The Irondequoit Creek basin, which encompasses parts of Monroe and Ontario Counties, has been the subject of many water-quality studies in response to public concern over the sedimentation and eutrophication of Irondequoit Bay (fig. 1). Irondequoit Creek is the bay's main tributary and its principal source of (1) nutrients that, in the past, supported algal blooms in the bay (Pixley, 1982); (2) dissolved chloride, which can alter the bay's chemical and thermal stratification and interfere with seasonal mixing of the bay's waters (Diment and others, 1974; Bubeck and Burton, 1989); and (3) sediment to which heavy metals and organic compounds can adhere (Schroeder, 1985).

Until the spring of 1978, effluent from 14 wastewater-treatment facilities within the basin and combined-sewer overflows (CSOs) from the city of Rochester was discharged to the creek. Since 1978, the wastewater-treatment effluent has been diverted out of the basin by a wastewater-interceptor system. In addition, a bedrock tunnel system that permits subsurface storage of stormwater has decreased the frequency of CSO discharges to the creek—from discharging during nearly every storm to discharging during only about one storm in 10 years (Michael Schifano, Monroe County Pure Waters, oral commun., 1999). Wastewater and combined-sewer water are now treated at a tertiary-treatment plant at the north end of Rochester and discharged into Lake Ontario.

Monroe County also has implemented several practices to decrease nonpoint-source chemical loads in Irondequoit Creek, including (1) onsite controls of erosion and storm runoff at new

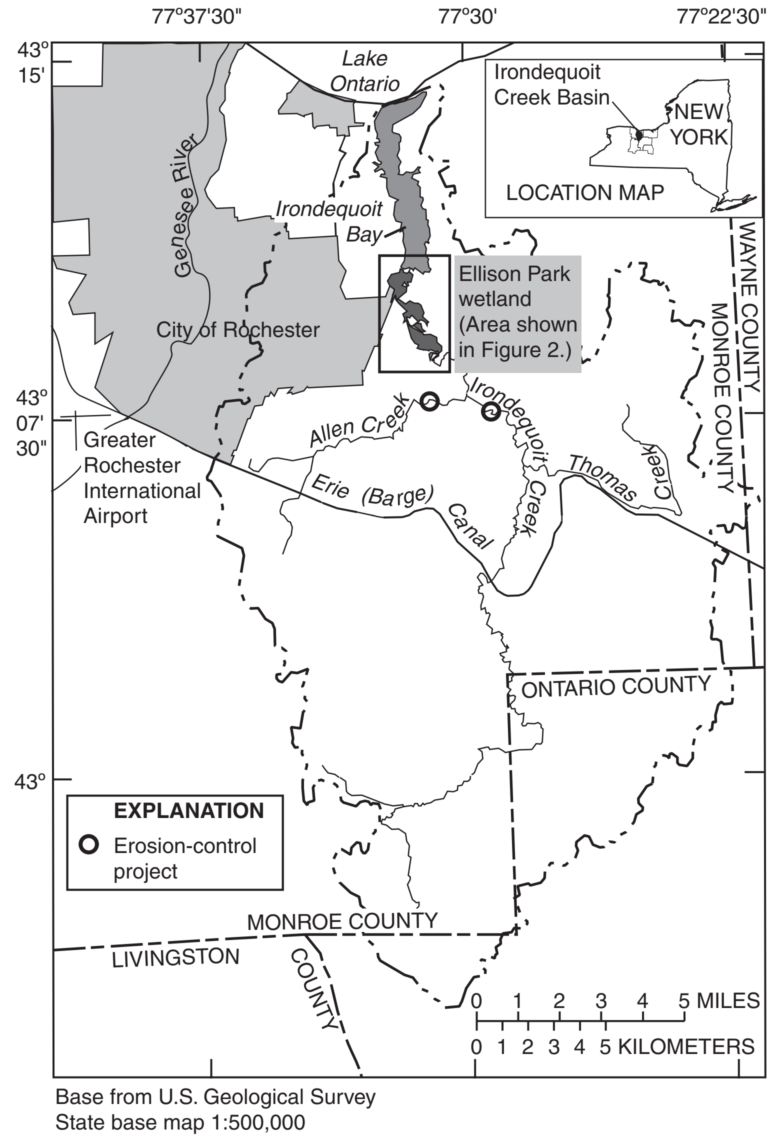


Figure 1. Location of study area and principal geographic features of Irondequoit Creek basin, Monroe County, N.Y.

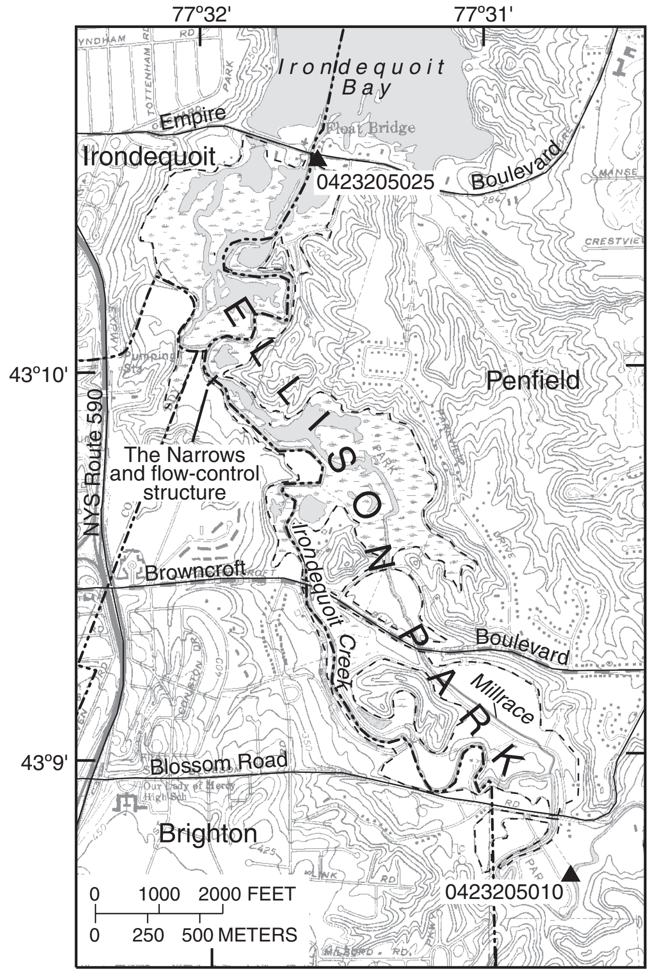
developments, with expanded use of multipurpose stormflow-detention basins; (2) conversion of dry detention basins in older developments to wet basins to promote the removal of nutrients in stormwater through increased biological uptake; and (3) rip-rap reinforcement of erodable banks along Irondequoit Creek and one of its tributaries, Allen Creek (fig. 1). In addition, the County has directly treated Irondequoit Bay through (1) application of alum to bottom sediments to inhibit the release of phosphorus to the water; and (2) injection of oxygen into the Bay's hypolimnion to maintain aerobic conditions. These measures, along with the diversion of wastewater out of the basin, have decreased nutrient loads to Irondequoit Bay (Coon and others, 2000) and have improved its eutrophic state (Monroe County Health Department, written commun., 2001), but further improvement has been slowed by the continued inflow of contaminants from nonpoint sources in the basin—primarily nutrients, metals, and organic compounds washed from impervious surfaces by rain and snowmelt (R.S. Burton, Monroe County Environmental Health Laboratory, written commun., 1996).

Previous Studies

Streamflow and water quality of Irondequoit Creek and its major tributaries, Thomas and Allen Creeks (fig. 1), have been extensively monitored by Monroe County and the U.S. Geological Survey (USGS). Irondequoit Creek basin was one of 28 basins that was included in the Nationwide Urban Runoff Program (NURP) during 1979-81. In addition, local projects have studied specific water-resource issues with the intent of extrapolating small-scale solutions to address basinwide concerns.

Irondequoit Creek Basin Studies

The hydrologic characteristics of the Irondequoit Creek basin were documented during 1979-81 as part of the NURP study (O'Brien and Gere, 1983; Zarriello and others, 1985; Kappel and others, 1986). One goal of the Irondequoit Creek NURP study was to assess the effect of storm runoff and its associated nutrients and contaminants on the quality of water in Irondequoit Bay. Kappel and others (1986) estimated annual loads of selected constituents that entered the bay (during 1980-81): 214 tons of ammonia-plus-organic nitrogen, 20.2 tons of total phosphorus, and 19,100 tons of total suspended solids. O'Brien and Gere (1983) estimated that 15.7 tons of phosphorus were entering Irondequoit Bay annually (1980-81) from external sources and that, given a zero net phosphorus release from the bottom sediments, the annual external loading would need to be decreased to about 5.6 tons to maintain the bay in a trophic state appropriate for recreational usage. They also indicated that the 3-month period that included the major seasonal snowmelt and spring runoff accounted for 50 to 75 percent of the annual phosphorus loads entering the bay.



Base from scanned U.S. Geological Survey Rochester East, NY 1:24,000, 1978

EXPLANATION

Wetland boundary

Town boundary

▲ 0423205010 Streamflow-measurement and water-quality monitoring site

Figure 2. Locations of streamflow- and water-quality monitoring sites in Ellison Park wetland, Monroe County, N.Y. (Location is shown in fig. 1)

A unique component of the Irondequoit Creek NURP study's monitoring program (Kappel and others, 1986) was the inclusion of an in-stream natural wetland (Ellison Park) (fig. 2). Several NURP projects considered wetlands to be promising mechanisms for treatment of urban runoff, but these projects did not include development of performance or design criteria (U.S. Environmental Protection Agency, 1983).

Historical (12,000 to 2,000 years before present), pre-European-settlement, and recent sedimentation rates in and upstream from Irondequoit Bay were estimated by Young (1992; 1996) and Schroeder (1985). Young (1992) used radiocarbon-dating techniques to estimate postglacial sedimentation rates from two sediment cores taken at the outlet of Irondequoit Bay (fig. 1) and one core taken from along Irondequoit Creek near Browncroft Boulevard (fig. 2); his results indicated the average historical sedimentation rate at the mouth of Irondequoit Creek to be 1 ft per 99 years, or 0.01 ft/yr (3 mm/yr). This rate is consistent with the rate of rise in the water-surface elevation, and consequent deepening, of the depositional pool along the southern shore of Lake Ontario in response to a differential rate of postglacial isostatic rebound of the Lake Ontario basin. The rate of glacial rebound at the northeastern end (outlet) of Lake Ontario is 1.25 ft per century and, at the southwestern end, is 0.25 ft per century (Clark and Persoage, 1970; Larsen, 1985). Because the lake outlet has been rising faster than the rest of the lake, the water level along the southern shore has been rising at a “differential” rate of about 1 ft per century. The rate of sediment accumulation in the vicinity of Irondequoit Bay, as computed by Young (1992), approximates this rate of water-level rise and the resulting deepening of the depositional pool in the area that encompasses the Ellison Park wetland and Irondequoit Bay.

Young (1996) estimated sedimentation rates for the past 100 to 1,200 years from the radiocarbon dates of 10 organic-matter samples from 6 excavation sites in Ellison Park between Blossom Road and Browncroft Boulevard (fig. 2). Results indicated an average sedimentation rate of 14.5 in. per century (0.012 ft/yr or 3.7 mm/yr) prior to European settlement of the Irondequoit Creek basin. This rate agrees closely with the historical sedimentation rate estimated by Young (1992) and the rate of isostatic rebound. Three samples from the post-settlement period indicate that sedimentation rates tripled—to an average rate of 43 in. per century or 0.036 ft/yr (11 mm/yr)—between the early 1800’s and 1950, presumably as a result of deforestation and land-use changes that accompanied settlement of the area. From a third to half of this rate is assumed to be a result of the southwestward tilting of the Lake Ontario basin in response to isostatic rebound; the remainder is attributed to increased erosion resulting from development within the Irondequoit Creek basin (Young, 1996).

Recent (1953-80) sedimentation rates estimated by Schroeder (1985) were based on lead-210 and cesium-137 radioisotopic analyses of four bottom-sediment cores taken from Irondequoit Bay in 1980; results indicated yearly sediment-accumulation rates of 0.10 to 0.36 g/cm^2 on a dry-weight basis. The calculated thickness of sediment deposited between 1964 (the year of peak cesium-137 concentration in the atmosphere) and 1980 (when the samples were collected) (data from R.A. Schroeder, U.S. Geological Survey, written commun., 1995), indicates a sedimentation rate of 0.0092 to 0.012 ft/yr (2.8 to 3.7 mm/yr), which is consistent with the historical rate estimated by Young (1992) but much less than the post-settlement rate estimated by Young (1996).

Ellison Park Wetland Studies

The Irondequoit Creek NURP study (O’Brien and Gere, 1983) identified the Ellison Park wetland as the most cost-effective means of decreasing nutrient loads entering Irondequoit Bay. Kappel and others (1986) estimated that the wetland decreased the phosphorus loads carried by Irondequoit Creek to Irondequoit Bay by 10 percent and theorized that an additional 15-percent decrease could be achieved, primarily through the deposition and mineralization of particulate phosphorus associated with the silt-clay sediment fraction, if flows through the wetland were regulated to increase water dispersal and detention time. Similarly, O’Brien and Gere (1983) estimated that the wetland could remove 24 percent of the creek’s phosphorus load if a flow-control structure were installed at the Narrows, a natural constriction midway through the wetland between Browncroft and Empire Boulevards (fig. 2).

An 11-year study of the wetland was begun in 1990 by the USGS in cooperation with the Monroe County Department of Health to evaluate the potential of the wetland to improve the water quality of Irondequoit Bay. During 1997, in response to the recommendation of O Brien and Gere (1983), Monroe County installed a flow-control structure (FCS) at the Narrows (fig. 2) to cause within-bank flows, especially the early spring first-flush and snowmelt flows, to overtop the banks of the main channel and disperse into the cattail-covered backwater areas of the wetland where natural detention, sedimentation, and biological uptake of nutrients could occur. This dispersal of stormflows over a larger area and for longer periods than occurred naturally would increase contact of the nutrient-laden water with plant life and microbes (bacteria, fungi, algae) and, theoretically, would decrease the loads of nutrients entering Irondequoit Bay and thereby improve its water quality.

Throughout the study, streamflow and water quality at the inflow and outflow of the wetland were monitored, and the flora and fauna in the wetland were surveyed periodically. (See “Reports of Biological Studies” section, at the end.) The study consisted of two phases: the first (1990-96) entailed an assessment of wetland functions and documentation of the wetland’s ecological status; the second (1997-2001, the subject of this report) entailed an assessment of changes in wetland functions or ecological status that might have resulted from the operation of the FCS.

The first phase of the study is summarized by Coon (1997) and Coon and others (2000). Coon (1997) described the hydrologic, sedimentological, and biological characteristics of the wetland and concluded the following:

1. Water levels in the wetland were controlled by the surface elevation of Lake Ontario.
2. Bankfull and lower flows of Irondequoit Creek did not disperse through the wetland, but were mostly confined within the banks of the main channel and usually passed through the wetland in less than 3.5 hours. Dispersal of stormflows occurred only when flows exceeded the capacity of the channel (overbank flows), which on average occurred twice a year.
3. Dispersed water moved into the cattail-covered backwater areas of the wetland, where it was detained for 3 to 15 hours or more (the maximum detention time was not measured).

4. Dispersal of flows also occurred during backwater conditions that resulted from high water levels on Lake Ontario, usually from April through July.
5. The channel sediments in the wetland ranged from sand, which was found in the main channel of Irondequoit Creek, to silt and clay with about 4-percent organic matter (by weight), which was found in the backwater areas.
6. Sediment accumulation in the wetland was sporadic, but an annual sedimentation rate of 0.006 to 0.016 ft/yr (1.8 to 4.9 mm/yr) was estimated.
7. The wetland, which was dominated by cattails (*Typha glauca*), was highly productive; estimated average total biomass was 5,230 g/m^2.
8. The wetland supported diverse fish and bird communities; at least 37 fish species and 28 possible wetland breeding-bird species were identified.

Coon and others (2000) assessed the wetland's effects on water quality of Irondequoit Creek and calculated inflow and outflow loads and removal efficiencies for selected constituents for 1990-96. The results indicated that the Ellison Park wetland was (1) a sink for total phosphorus and total suspended solids—28 and 47 percent removal efficiencies, respectively; and (2) a source of orthophosphate and ammonia— -38 and -84 percent removal efficiencies, respectively. The wetland facilitated a slight decrease in total nitrogen loads and significant decreases in concentrations of zinc, iron, and lead, but had a negligible effect on conservative constituents such as chloride and sulfate.

Purpose and Scope

This report presents the results of the second phase of the study (1997-2001), which entailed an assessment of the wetland water levels, biota, sedimentation rates, and water and sediment quality during the first 4.5 years after installation of the FCS, and comparison of these data with those collected during the precontrol period (1990-96). The increased flooding potential was evaluated from discharge measurements and water-surface profiles. The plants, fish, benthic-macroinvertebrate, and bird communities were evaluated through biological surveys. Sedimentation rates were measured, and sediment samples were analyzed for major and trace elements, polycyclic aromatic hydrocarbons, and organochlorine and organophosphate compounds. Concentrations of selected constituents were measured in inflow and outflow water samples, and loads of these constituents, as well as the wetland's removal efficiency for each, were calculated. This report (1) summarizes the design of the FCS and the ecological-monitoring program; (2) describes the methods used for each component of the study—water-level and discharge measurements, ecological surveys, sediment studies, water-quality analyses, estimation of monthly chemical inflow and outflow loads, and calculation of the wetland's removal efficiency for selected constituents; (3) presents results of each of these study components; and (4) assesses the changes in wetland functions and characteristics that might be attributable to the FCS. Monthly and annual inflow and outflow loads and the estimated removal-efficiency values for phosphorus compounds, nitrogen compounds, suspended sediment, chloride, and sulfate are presented in an appendix.

Acknowledgments

This project required the combined efforts of many people. The Monroe County Environmental Health Laboratory (MCEHL) assisted in the maintenance and operation of the streamflow- and water-quality-monitoring sites, analyzed water samples, participated in several components of the ecological-monitoring programs, and operated and maintained the FCS at the Narrows. Richard Burton, Anna Madden, Charles Knauf, and Gary Brown (MCEHL laboratory and project administrators) provided guidance and suggestions throughout the project. The FCS was designed by Mark Ballerstein of the Monroe County Department of Environmental Services, and was installed by C.P. Ward, Inc., under the oversight of William Keihl, Harold Ford, and Randall Williamson. Access to the Narrows for installation and operation of the FCS was granted by the Town of Brighton.

The plant surveys were conducted by Dr. John Bernard, Department of Biology, Ithaca College; Dr. Franz Seischab, Department of Biology, Rochester Institute of Technology; and Dr. John Hunter (Department of Biological Sciences) and Dr. Mark Noll (Department of Earth Sciences), State University of New York (SUNY) College at Brockport. Field and laboratory assistance for the plant surveys was provided by Nicole Perry, Jamie Brennan, Duane Hanselman, Tracy Cohen, Jessie DiLorenzo, Amy Gardner, Heather Halbritter, Jamie Kirby, Kit Sheehan, Frank Evans, Nicholas Byerle, Sarah Lewis, and Laura Sontag-Brown.

The fish surveys were conducted by Dr. Neil Ringler, Daniel Miller, Darran Crabtree, and Patricia Thompson (Department of Environmental and Forest Biology), SUNY College of Environmental Science and Forestry at Syracuse.

Benthic macroinvertebrate work was conducted by Dr. James Haynes (Department of Biological Sciences) and Dr. James McNamara (Department of Mathematics), SUNY College at Brockport.

Bird surveys were conducted by volunteers Robert McKinney and Robert Spahn, members of the Genesee Ornithological Society, Rochester, N.Y.

Data on (1) the wetland's ground-water hydrology and chemistry; (2) the movement, deposition, and generation of phosphorus in the wetland; and (3) evapotranspiration by the cattails and the effect of this process on surface water were collected by Cornell University graduate students Michael Traynor and Oliver Pierson, under the supervision of Dr. Rebecca Schneider, Department of Natural Resources, Cornell University, Ithaca, N.Y.

STUDY AREA

The Ellison Park wetland occupies 423 acres at the mouth of Irondequoit Creek, which drains 151 mi^2 and flows northward into Irondequoit Bay (fig. 1). On the basis of satellite imagery (New York State, 2000), only 25 percent of the Irondequoit Creek basin is classified as "developed," but tax-parcel data classify about 54 percent of the basin as urban or suburban (Christopher Sciacca, Monroe County Planning Board, written commun., 1998); most of this developed area is north of the New York State Erie (Barge) Canal (fig. 1). The upstream (southern) part of the basin is dominated by forest and farmland but is becoming increasingly developed.

Average annual precipitation in the Rochester area is 32.4 in. (168 years of record), including the amount derived from an average annual snowfall of about 90 in. (55 years of record), as recorded by the National Weather Service station at the Greater Rochester International Airport (National Climatic Data Center, 1996). Monthly precipitation and the resultant monthly discharges during 1990-2001 (Northeast Regional Climate Center, Ithaca, N.Y., written commun.) are plotted with 30-year (1961-90) mean monthly precipitation values in figure 3A and with monthly maximum, mean, and minimum discharges for the period of record (1981-2001) for Irondequoit Creek above Blossom Road in figure 3B.

The wetland lies along the township boundaries of Irondequoit, Brighton, and Penfield in Monroe County and is mostly within Ellison Park, a county park (fig. 2). The wetland, which forms a transition zone between the riparian environment of the creek and the lacustrine environment of Irondequoit Bay at and below an elevation of 250 ft (DeGaspari and Bannister, 1983), represents less than 0.5 percent of the creek's total drainage area. Bounded by steep valley sides on the east and west, the wetland covers the entire valley floor from its inlet above Blossom Road to Irondequoit Bay at Empire Boulevard (fig. 2). The area between Browncroft and Empire Boulevards is dominated by cattails and is divided by a constriction at the Narrows into a southern and a northern segment (263 acres and 160 acres, respectively, DeGaspari and Bannister, 1983). The width of the wetland ranges from 200 ft at the Narrows to 2,500 ft in the southern and northern parts.

The wetland is classified as a "palustrine persistent emergent" wetland by the U.S. Fish and Wildlife Service (Cowardin and others, 1979); that is, a nontidal marsh characterized by erect, rooted, herbaceous hydrophytes that may be temporarily or permanently flooded at the base and normally remain standing until at least the beginning of the next growing season. Cattails, primarily *Typha glauca*, cover about 265 acres, or 63 percent of the wetland area between Browncroft and Empire Boulevards. (Acreages were digitized from a USGS 7.5-minute topographic map, Rochester East, N.Y. quadrangle.) Plant species richness is greatest on the banks of the main channel of Irondequoit Creek and along the wetland margins, where soil wetness varies greatly with slight differences in elevation. Open water, including channels, covers about 99 acres, or 23 percent of the wetland. A forested wetland area between Browncroft Boulevard and Blossom Road covers about 25 acres, or 6 percent of the wetland, and maintained grass areas of the county park below the wetland-demarcation elevation of 250 ft (National Geodetic Vertical Datum, NGVD) cover the rest of the wetland (about 34 acres, or 8 percent of the wetland).

The wetland contains hydric soil, defined by the U.S. Soil Conservation Service (1985) as soil that, in its undrained condition, is saturated, flooded, or ponded long enough during the growing season to develop anaerobic conditions that favor the growth and regeneration of hydrophytic vegetation. The wetland soil is a mineral soil, formed through the deposition of alluvium (flood-plain deposits) from the Irondequoit Creek basin, that contains partly decomposed organic material. Organic matter in soil samples from the cattail-covered areas of the wetland represented from 35 to 40 percent of the sample material (by weight) on average, but represented as much as 79 percent of some samples (Hunter and others, 2000; Noll and others, 2002).

Wetland conditions in Ellison Park are dependent on the surface elevation of Lake Ontario, which is maintained within a narrow range by control structures on the St. Lawrence Seaway. Stage data from a streamflow-monitoring station at Empire Boulevard (fig. 2) indicated that water-surface elevations at the mouth of the wetland fluctuated less than 4.5 ft during 1990-2001 (fig. 4). Water levels in Lake Ontario usually rise during the spring-runoff period and are maintained through the summer for navigational and recreational purposes and for hydroelectric-power generation on the St. Lawrence River near Massena, N.Y. Water levels are lowered in December and January to increase the storage capacity for spring runoff (Paul Yu, U.S. Army Corps of Engineers, oral commun., 1997). A water-surface elevation of 246.9 ft at Empire Boulevard results in the inundation of most of the cattail-covered area throughout the wetland. A drop of less than 0.5 ft causes water levels in the southern wetland area to fall below land surface, leaving only the northernmost part of the northern wetland area inundated. Water levels throughout the wetland range from at least 2 ft above land surface to 1.5 ft below. Flat, vegetated areas of the wetland can be briefly inundated by runoff from large storms and are seasonally inundated for periods of 4 to 7 months. The duration of seasonal inundation (hydroperiod) varies from year to year and increases from south to north; in some years, the water level rises only enough to inundate the southern wetland area for several days, whereas in other years, high water levels can persist from January through August (fig. 4; Coon and others, 2000).

Two channels convey water through the wetland—the main channel of Irondequoit Creek, which carries most of the flow, and a smaller channel, locally referred to as the Millrace (fig. 2), which formerly was a saw-mill raceway on the east bank of the creek just downstream from Blossom Road (F.W. Pugsley, Town of Pittsford Historian, written commun., 1942); currently it is a diversion from the main channel to the eastern part of the southern wetland area north of Browncroft Boulevard. The main channel in the Blossom Road vicinity is about 70 ft wide and 10 to 12 ft deep at bankfull stage, whereas the Millrace is 20 to 25 ft wide and 5 to 6 ft deep. The main-channel dimensions decrease between Browncroft Boulevard and the Narrows to a bankfull width of 50 to 60 ft and depth of 5 to 8 ft. The extension of the Millrace channel into the southern wetland area widens to about 50 ft but is only about 3 ft deep. The depth of Irondequoit Creek downstream from the Narrows gradually decreases to 3 or 4 ft.

STUDY DESIGN

The FCS was installed in 1997 at the Narrows, where the valley-bottom width is about 200 ft and the channel width is about 60 ft. The Narrows divides the main cattail-covered area of the wetland into a 144-acre southern (upstream) area between the Narrows and Browncroft Boulevard and a 160-acre northern

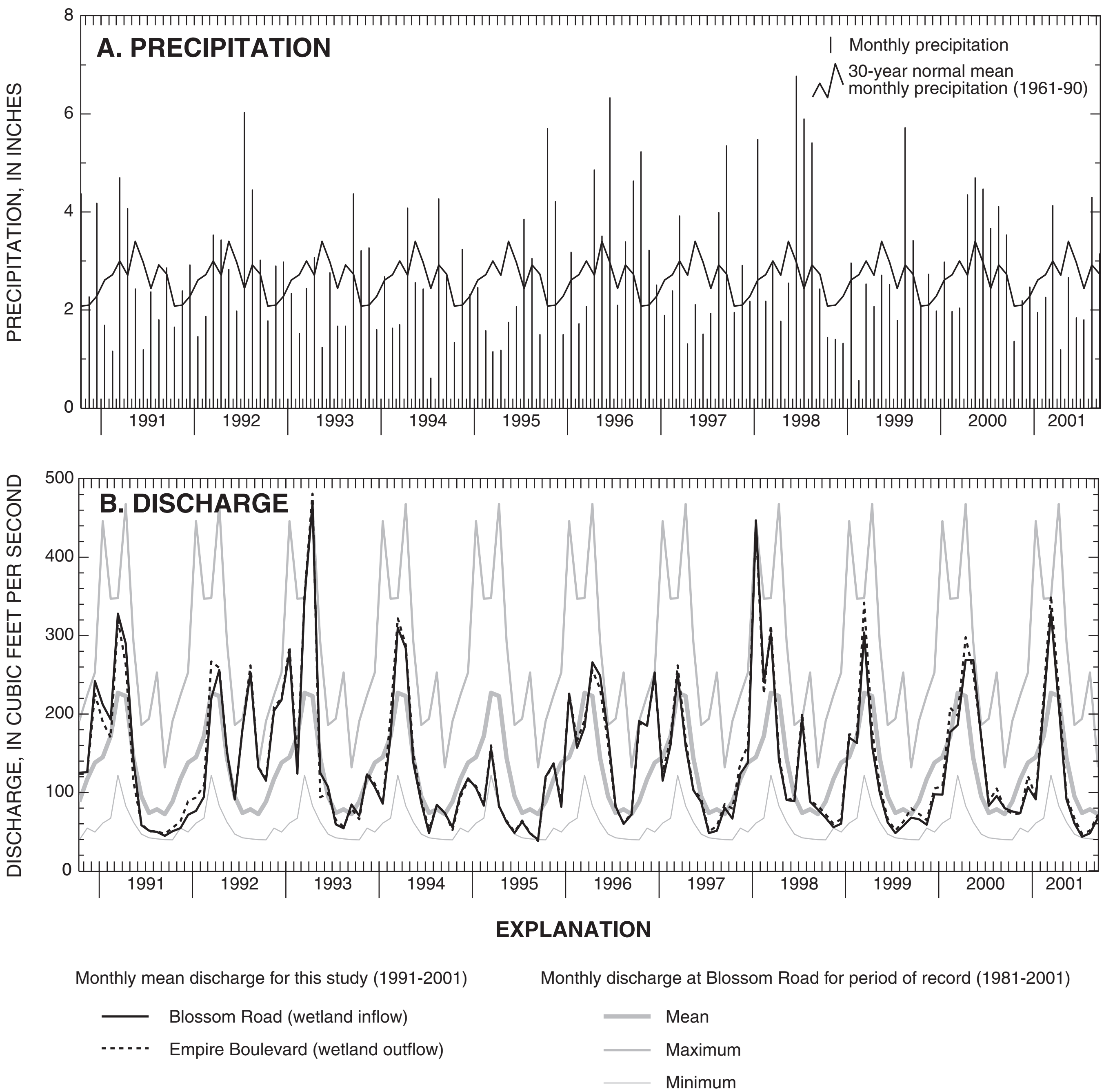


Figure 3 . A. Monthly precipitation (1990-2001) and normal mean monthly precipitation (1961-90) recorded at Greater Rochester International Airport, Rochester, N.Y. B. Monthly mean discharges of Irondequoit Creek above Blossom Road and at Empire Boulevard, Monroe County, N.Y. (1990-2001), with monthly maximum, mean, and minimum discharges for period of record at Blossom Road (1981-2001) (Locations are shown in figs. 1 and 2.)

(downstream) area between the Narrows and the wetland outlet (Empire Boulevard) (fig. 2). As explained in the previous section, water depths in the northern area are greater, and periods of inundation persist longer than in the southern area. Comparison of periodic measurements of water depth in the cattail-covered areas with water-surface elevations at the wetland outlet indicates that much of the cattail mat in the northern area floats when water levels are high, whereas much of the cattail mat in the southern area is anchored to the underlying sediment.

The entire wetland has been subjected to large chemical loads from nonpoint-source urban runoff for decades and, until 1978, received effluent from wastewater-treatment plants and combined-sewer overflows. The northern area also received chemical loads from two other sources—a landfill (currently closed to household wastes but used for composting operations) and a combined-sewer overflow (whose overflow frequency was decreased from almost every storm in the 1970's to less than once in 10 years since the mid-1980's). Both sources are on the western side of the wetland and close to the Narrows.

These differences were considered minor when the two wetland areas were assessed for suitability for data comparisons. Other nearby wetlands along Lake Ontario were considered as control sites for this study but were rejected because of substantial differences, despite their similar locations in the landscape.

Flow-Control Structure

The FCS, which was designed and operated by Monroe County, consisted of two parts—a semipermanent base control made of stop logs and sheet metal, and a dynamic control made of hinged gates and removable stop logs (fig. 5). The base control was designed to (1) approximate the channel cross-section shape at a point about 100 ft downstream from the Narrows where an erosion-resistant till creates a shallow, gravel riffle in the creek during low-water periods, and (2) have minimal effect on flood levels upstream from the FCS. This latter criterion was based on the results of hydraulic analyses of high flows and water-surface elevations with a one-dimensional, gradually varied, steady-flow model (Shearman, 1990). The dynamic control was designed and operated to have minimal effect on flooding in the day-use area of the park between Blossom Road and Browncroft Boulevard and to have no effect on flood levels at and above Blossom Road, about 2.4 mi upstream (fig. 2).

Construction of the base control entailed driving nine H piles, spaced about 6 ft apart, into the channel bottom to the point of resistance. Foot-square stop logs were slid into the channels of adjacent piles to an elevation of 244 ft in the four center bays and 246 ft in the side bays, which created a 2-ft deep notch in the center of the structure. The bottom elevation of the notch was below the thalweg elevation of the riffle downstream to ensure that the notch would be submerged at all times and would permit unobstructed passage of fish and small boats during low-water periods.

The dynamic control, which permitted water-level manipulation to an elevation of 247 ft, was built upon the base control and consisted of removable stop logs above the side sections of the FCS and hinged gates in the "notch" bays (fig. 5). Operational needs and public concerns warranted modifications to this design, which included (1) driving sheet-metal plates into the channel bottom along the upstream face of the stop logs to prevent undercutting of the structure; (2) cutting a center H-pile to increase the maximum opening between piles

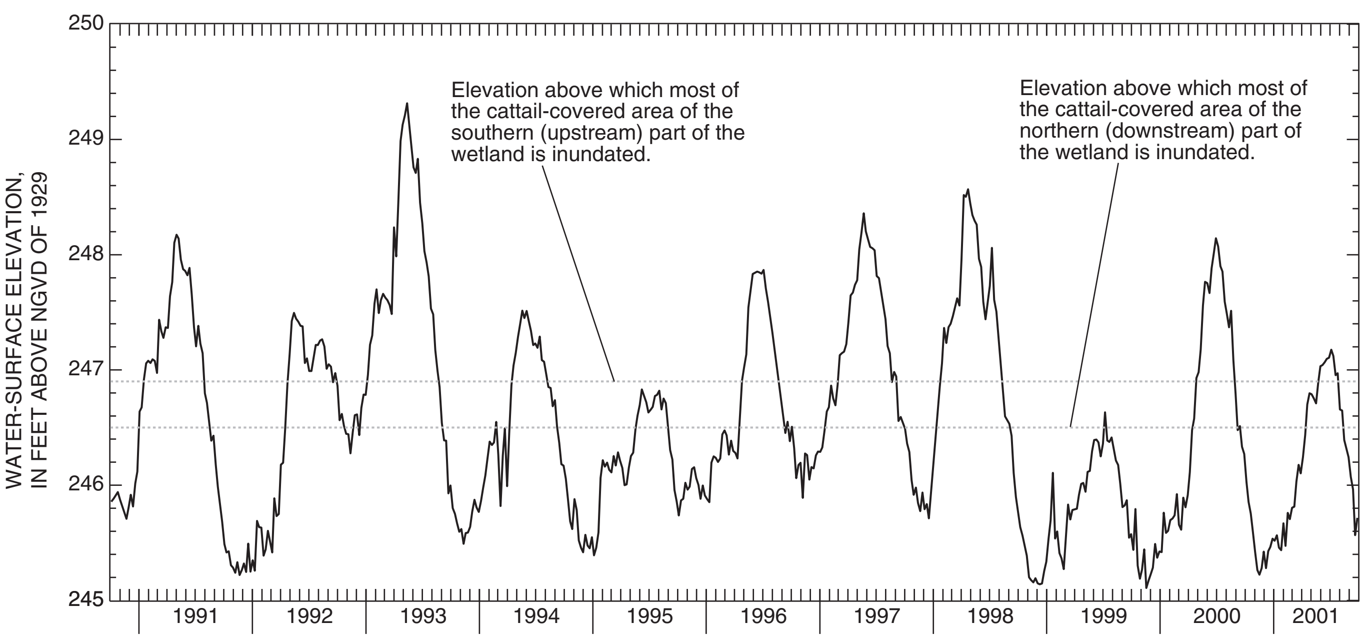


Figure 4. Daily mean water-surface elevation of Ellison Park wetland, Monroe County, N.Y., as measured at wetland outlet (Empire Boulevard), 1990-2001. (Location is shown in fig. 2.)

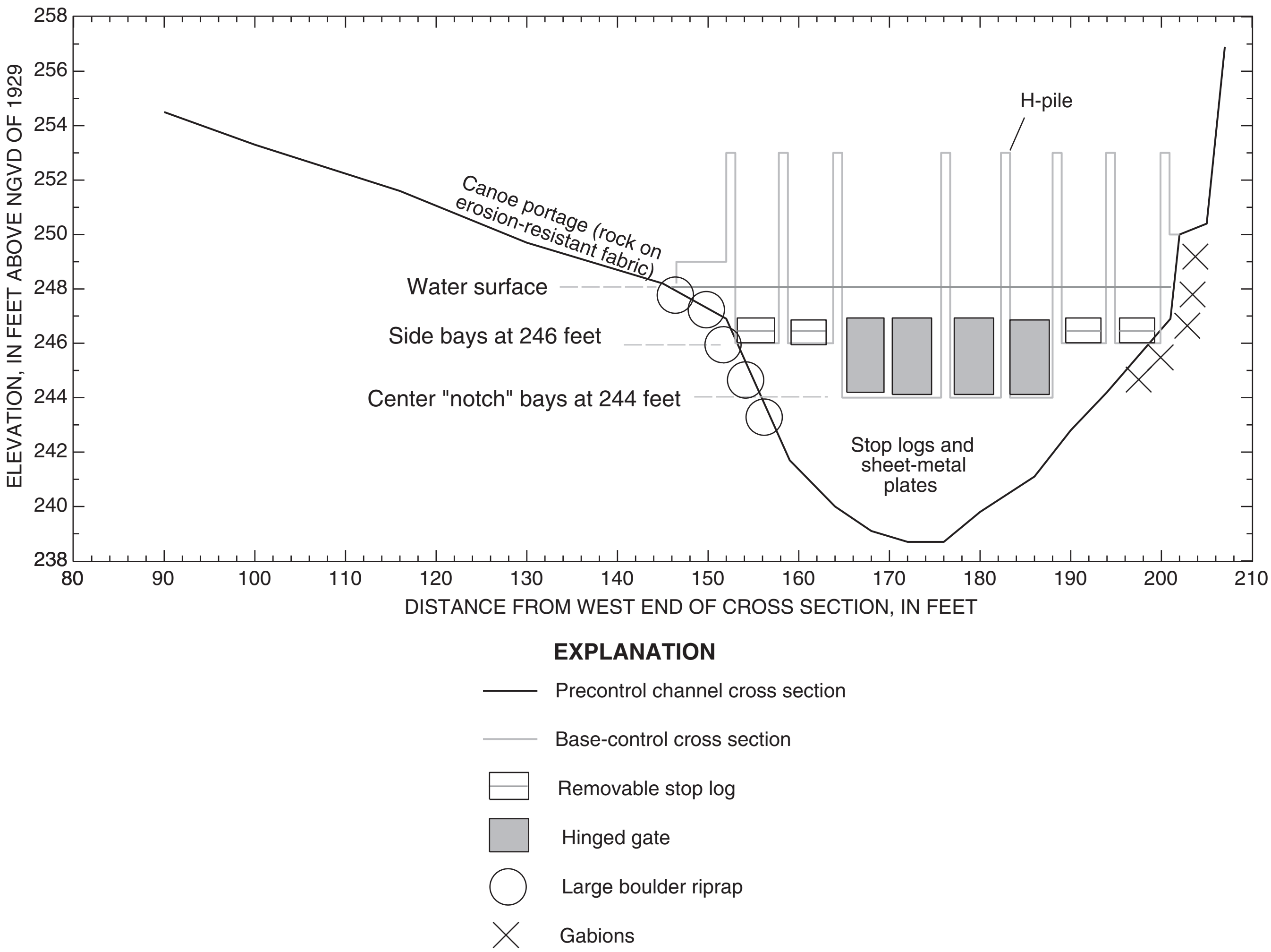


Figure 5. Schematic diagram and photograph of flow-control structure at the Narrows in Ellison Park wetland, Monroe County, N.Y., 2001. A. Schematic diagram. B. Downstream-facing view of flow-control structure with H-piles extending above water surface.

to 11 ft to facilitate passage of canoes and small boats; (3) laying large boulder riprap or gabions on the streambanks, and erosion-resistant fabric and rock on the west-bank overflow area to control erosion; and (4) constructing a canoe portage around the FCS along the western bank.

The purpose of the FCS was to increase the dispersal and short-term detention of stormflows with recurrence intervals between 1 and 2 years—the annual peak flow. Under natural conditions, stormflows of this magnitude are confined between the elevated banks of the creek channel, and little water flows into the cattail-covered areas of the wetland (Coon, 1997). Manipulation of the FCS could cause the water-surface elevation associated with these frequent high flows to rise above bank level and thereby increase the dispersal of flows into the cattail-covered areas. It also could cause larger flows—those that would ordinarily overtop the banks—to disperse more widely, and to be detained longer, than would occur without the FCS. Detention time of stormflows in the southern part of the wetland was expected to increase by as much as 0.7 days (17 h) for small peak flows (500 ft^3/s); from 2.5 to 4.8 days (60 to 115 h) for high peak flows (1,500 ft^3/s); and from 2.6 to 4.7 days (62 to 113 h) for extremely high peak flows (4,000 ft^3/s) (Kappel and others, 1986). The actual period of inundation depends not only on the magnitude of flow, but on the antecedent water levels in Irondequoit Bay and Lake Ontario.

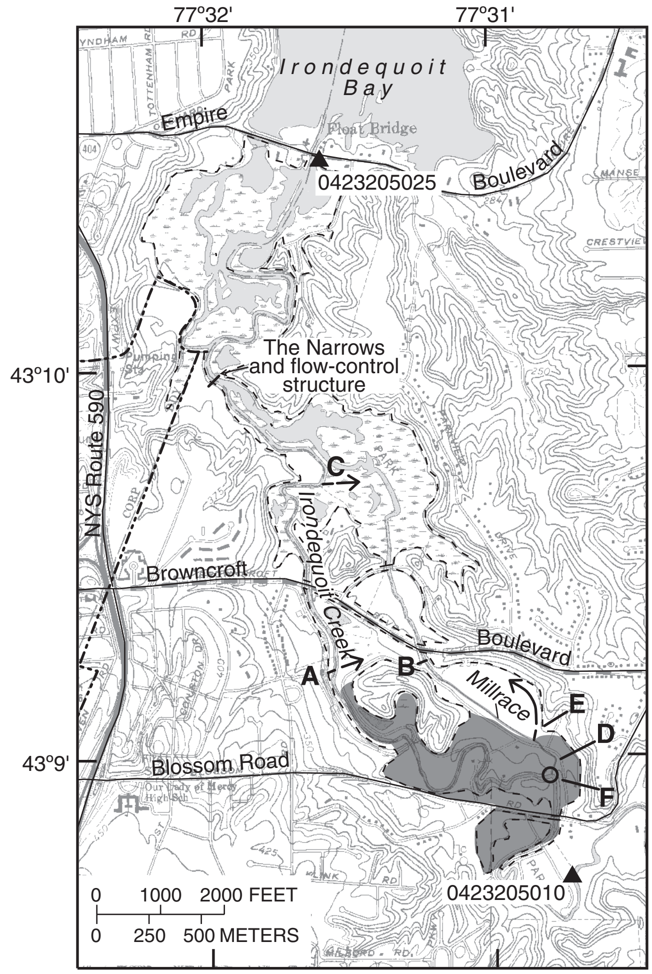
The FCS was operated in accordance with weather forecasts and water levels at the Blossom Road streamflow-monitoring station. The gates in the “notch” bays adjacent to the center bay and the removable stop logs were left in their maximum-detention positions most of the time. In anticipation of stormflows, the Monroe County Environmental Health Laboratory (MCEHL) would raise the gates in the center bay. If stormflows remained within the creek’s banks at Blossom Road, no adjustments were made to the FCS, but if weather forecasts predicted additional rain that might cause flooding at Blossom Road, the removable stop logs were removed, and the gates were lowered to obtain the base-control configuration.

The FCS was designed and operated to permit free passage of fish under all flow conditions and downstream water levels. During the period of seasonally high water levels (late spring and summer), the FCS was submerged, and water flowed over its top. During low-water periods (late fall and winter), water flowed through the 11-ft-wide notch in the center of the FCS at all times to a depth of at least 2 ft.

The base control was installed near the end of February 1997, and the gates in the center notch were installed in early September 1997. The FCS was fully operational by September 15, 1997. An operation record was maintained, in which changes in stop-log placement and position of gates were noted.

Additional Flow Modifications

Natural flow detention that occurs in the southeastern part of the wetland, and the potential water-quality benefits that might be realized if water from Irondequoit Creek were diverted to this cattail-covered backwater area, have been described by Coon (1997) and Coon and others (2000). To maximize these benefits, the flow patterns in the creek and Millrace were modified by Monroe County between February 1997 and the



Base from scanned U.S. Geological Survey
Rochester East, NY 1:24,000, 1978

EXPLANATION

- - - - - - - - Wetland boundary

▲ 0423205010 Streamflow-measurement and water-quality monitoring site

Direction of flow through streambank notch

Day-use area

Letter indicates location and type of hydrologic modification

-A,B,C Streambank notch

D Channel widening

E Diversion channel

○ F Culvert

Figure 6. Locations of flow modifications to Irondequoit Creek and the Millrace, Ellison Park wetland, Monroe County, N.Y.,1997-98.

summer of 1998 to (1) increase dispersal of flows along the Millrace and in the wetland, (2) decrease flooding between Blossom Road and Browncroft Boulevard (the day-use area of the park), and (3) divert as much as 50 percent of bankfull stormflows from the creek to the Millrace, which drains directly into the southeastern part of the wetland (fig. 6). First, the bank of the creek was notched upstream from Browncroft Boulevard (point A in fig. 6) to increase diversion of within-bank flows to the wetland area bounded by Browncroft Boulevard to the north, the creek to the west, and the Millrace to the east. A second notch was made at the Millrace end of this wetland area (point B) to allow water that exceeded the storage capacity of this area to pass into the Millrace. A third notch was constructed in the east bank of Irondequoit Creek within the wetland proper, about halfway between Browncroft Boulevard and the Narrows, to permit movement of water from the creek directly into the backwater area of the wetland (point C). Additional modifications to the Millrace included (1) widening the channel immediately downstream from the point of diversion from Irondequoit Creek (point D), and (2) constructing a top-of-bank diversion channel from the Millrace to a wooded wetland area east of the Millrace and south of Browncroft Boulevard (point E). Finally, the top of bank was lowered and a third culvert was added to the existing two culverts at the point of diversion from Irondequoit Creek to the Millrace just downstream from Blossom Road to increase flow to the Millrace (point F).

Ecological Monitoring Program

The ecological-monitoring program was designed to identify any effects that the FCS might have on the wetland's ecology. The program was based on the advice of a technical oversight committee that included representatives of the USGS, U.S. Army Corps of Engineers, U.S. Fish and Wildlife Service, New York State Department of Environmental Conservation, county and town agencies, and researchers from biology and geology departments of local universities. The program included periodic surveys of plants, fish, benthic macroinvertebrates, and birds. The plant surveys were designed to (1) identify plant species, (2) map plant communities, (3) measure cattail density and biomass, and (4) analyze above- and below-ground cattail tissues along permanent transects across the wetland. Fish surveys were designed to (1) identify fish species, (2) describe community composition, (3) record use of the wetland by resident and nonresident fish, (4) document species distribution within the wetland, and (5) note symptoms of contaminant-related stress. An unplanned consequence of using trapnets during the fish surveys was the capture of turtles, which were counted and identified by species. The benthic-macroinvertebrate and bird surveys were designed to (1) document species richness, and (2) confirm the presence or absence of species. Each of these studies included assessment of any identifiable or probable effect that the FCS had or might have on the particular biological community that was surveyed. The monitoring program also entailed (1) annual measurement of sediment accumulation at permanent measurement points; and (2) periodic collection and analysis of sediment samples for major and trace elements and organic compounds, including pesticides, polychlorinated biphenyls, and polycyclic aromatic hydrocarbons. Baseline surveys and measurements were made during 1991-96 to document the precontrol differences between the southern and northern wetland areas. These results were compared with those obtained from similar surveys and measurements conducted during the postcontrol period to identify changes that might be attributable to the FCS. The timing and frequency of the components of the ecological-monitoring program during the study period are given in table 1.

METHODS

The Ellison Park wetland project consisted of several studies to assess changes in the wetland after 1997 that might be attributable to the FCS. These were: (1) studies to assess the effects of the FCS on flooding extent and high-water levels; (2) an ecological-monitoring program that consisted of periodic surveys of the plant, fish, benthic-macroinvertebrate, and bird communities; (3) calculation of sedimentation rates, and chemical analyses of sediment samples; (4) measurement of discharge and water quality at the inflow and outflow of the wetland; (5) a quality-assurance and quality-control program to assess the accuracy and reliability of the water-quality data, which were provided by the MCEHL; and (6) calculation of the loads of selected chemical constituents in wetland inflow and outflow to obtain removal-efficiency values. The methods used in each of these study components are described below.

Water-Surface-Elevation and Discharge Measurements

Two studies were conducted during the postcontrol period to assess the effect of the FCS on flooding and high-water levels, especially in the day-use area of Ellison Park in the vicinity of Blossom Road (fig. 6). The first study entailed use of WSPRO, a water-surface-profile model of one-dimensional, gradually varied, steady flow in open channels (Shearman, 1990). The model was calibrated from measured water-surface elevations and discharges along Irondequoit Creek and the Millrace from the Narrows to Blossom Road, and was then used to predict the water-surface profiles that would result from the FCS during high flows. Water-surface elevations were measured during major storms with a weighted steel tape from surveyed measuring points on six bridges that span Irondequoit Creek and the Millrace. These elevations were compared with the water-surface profiles generated by WSPRO to confirm the extent of backwater from the FCS.

The second study entailed making high-flow measurements at Blossom Road, about 13,000 ft upstream from

Table 1. Components of ecological-monitoring program for Ellison Park wetland study, Monroe County, N.Y., 1991-2002, by year

Component	1991	1992	1993	1994	1995	1996	1997[a]	1998	1999	2000	2001	2002
Plant survey												
Identify plant species, map plant communities, measure cattail density and biomass, analyze above- and below-ground cattail tissues along permanent transects across wetland	X	-	-	-	-	X	-	-	X	X[b]	X	-
Fish survey												
Document species diversity and community composition, use of wetland by resident and nonresident fish, species distribution, and symptoms of contaminant-related stress	X	-	-	-	-	X	-	-	X	-	X	-
Benthic-macroinvertebrate survey												
Document species diversity and distribution, confirm presence or absence of selected species	X	-	-	-	-	X	-	-	-	-	-	-
Bird survey												
Document species diversity, confirm presence or absence of selected species	X	-	-	-	-	X	-	-	-	-	-	X
Sedimentation measurement												
Measure accumulation at permanent measurement points	-	X	X	X	X	X	X	X	X	X	X	X
Sediment-quality analysis												
Periodic collection and analysis of samples for major and trace elements, and organic compounds, including pesticides, polychlorinated biphenyls, and polycyclic aromatic hyrdocarbons	-	-	-	X	-	-	X	-	X	-	X	-

[a] Flow-control structure was installed February 1997; gates in center notch were installed September 1997.

[b] Chemical analyses of cattail tissues and calculation of chemical standing stocks were not done.

the FCS, and checking against the stage-to-discharge relation for the USGS streamflow-monitoring station above Blossom Road to identify any measurable backwater that could be attributed to the FCS at that point. The stage-to-discharge relation was based on 15 years (1981-96) of discharge measurements. Discharge was measured by standard USGS current-meter methods as described by Rantz and others (1982).

Plant Surveys

Five wetland-plant surveys were conducted. The first two (1991 and 1996) preceded the FCS installation and were done by a research team from the Departments of Biology at Ithaca College, Ithaca, N.Y., and Rochester Institute of Technology, Rochester, N.Y. (Bernard and Seischab, 1991; 1997; Bernard and others, 1998). The last three surveys (1999-2001) were done by the State University of New York (SUNY) College at Brockport (Hunter and others, 2000; Noll and others, 2002).

The survey design was based on the hypothesis that the FCS might cause prolonged periods of higher water levels in the southern wetland area than typically occurred prior to FCS installation and could thereby (1) decrease the density and growth of cattails and decrease plant-species richness; or (2) alter the chemical composition of cattail tissue upstream from the FCS through preferential deposition of sediment. The surveys consisted of (1) identification of individual plant species, (2) mapping of plant communities, (3) measurements of cattail density and above- and below-ground biomass, and (4) chemical analyses of above- and below-ground cattail tissues. The 1991 plant identification and mapping, and measurements of cattail density, were conducted on June 5 and July 5; cattail tissues for biomass calculations and chemical analyses were collected from August 15 to September 10. All components of the 1996 study were conducted from July 25 to August 25. All work for the 1999, 2000, and 2001 surveys was done during August 14-18 of each year. The 2000 survey was done gratuitously by the SUNY Brockport researchers and included only plant identification and measurements of cattail density and biomass.

Two sampling transects (fig. 7) were established—one across the southern part of the wetland, and one across the northern part—along which 3.3 x 3.3-ft (1-m^2) permanent plots were established every 33 ft (10 m); the northeastern corner of each plot was marked with a 1.5-in. PVC pipe. A third transect was established in 1996 in a cove in the southeastern part of the wetland that had been identified in 1991 as an area of unique

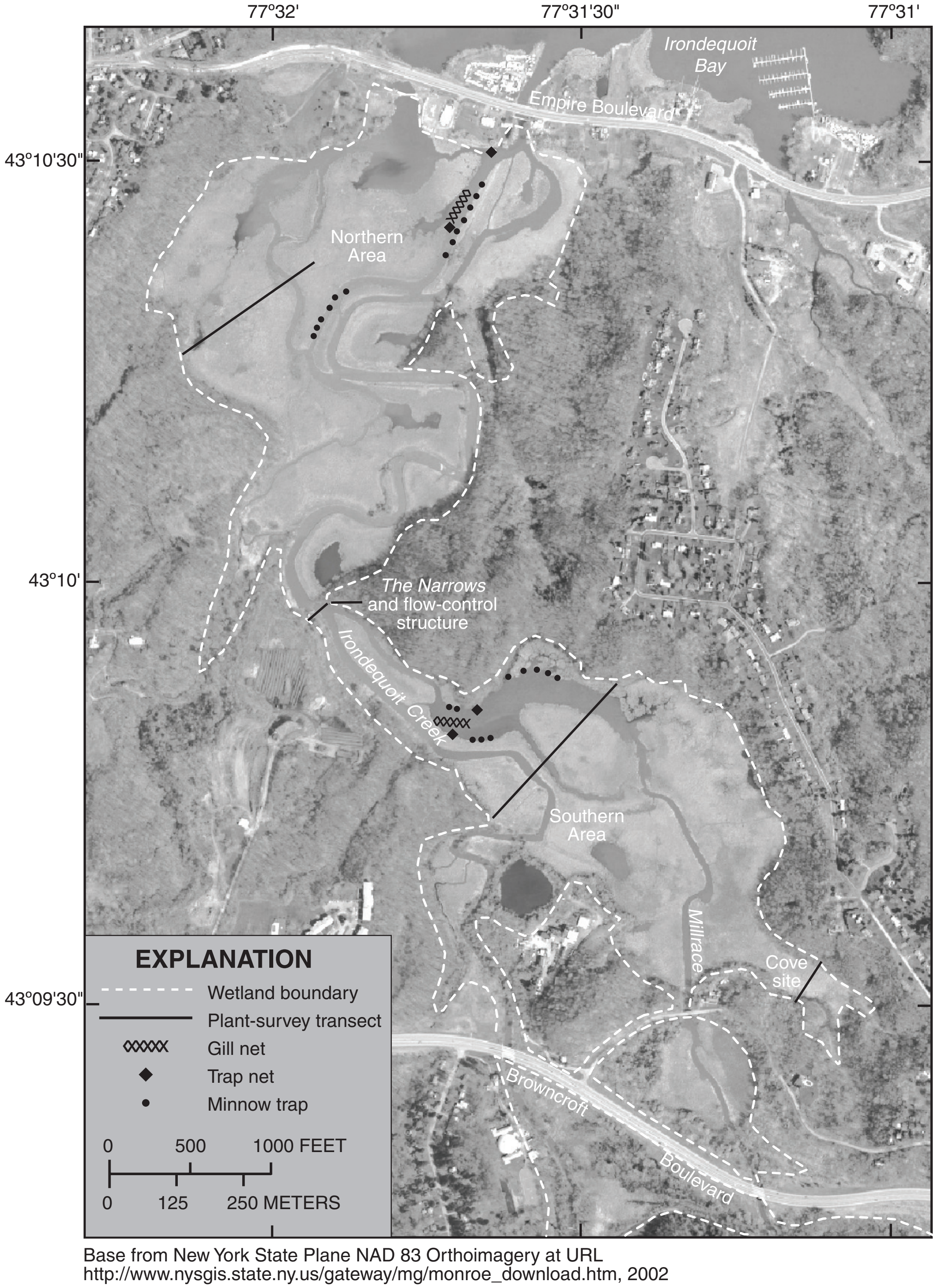


Figure 7. Locations of data-collection sites for plant and fish surveys in Ellison Park wetland, Monroe County, N.Y., 1991-2001

plant-species richness. Species nomenclature was in accordance with Gleason and Cronquist (1991), except for *Typha glauca*, which in this study was treated as a species, whereas Gleason and Cronquist (1991) treat it as a hybrid. The height of cattail shoots, cattail density, percentage of plot surface covered by cattails, and water depth were measured at each plot.

Additional plots were established in 1999 along the elevated banks (levees) of the main channel of Irondequoit Creek, where a relatively large wetness gradient—from saturated to dry conditions—extends along a relatively narrow strip of land and permits the growth of a more diverse plant community than is found in the low-lying, cattail-dominated areas nearby, where wetness conditions are uniform, at least seasonally. Aquatic plots in the channels adjacent to the levee plots were established to identify submerged and floating plants. These plots were surveyed during 1999 and 2001 to document species richness.

Methods for biomass collection and processing, and for chemical analysis of the cattail-tissue samples varied, depending on the contractor. Every effort was made to minimize these differences to ensure comparability of the analytical results.

1991 and 1996 Protocols

Sample collection and biomass calculation. During the 1991 survey, samples of above-ground tissue were harvested from a 3.3 x 3.3-ft (1-m^2) area adjacent to each permanent plot, and samples of below-ground tissue were collected from a 0.82 x 0.82-ft (0.25 x 0.25-m) area within each of the adjacent plots to a depth of about 12 in. (30 cm). Each sample was washed free of sediment and debris, then air-dried and weighed, and the biomass per unit area was computed. Unusually high water levels and hazardous walking conditions in the field, as well as a need to decrease the time needed to process the large volume of plant material, prompted modification of the above procedures for biomass collection across the northern transect during the 1991 survey. This modification entailed collection of a representative sample of above-ground biomass (2 or 3 cattail shoots) adjacent to each permanent plot, and use of their rhizomes and roots as the below-ground biomass sample. After drying and weighing, the above-ground biomass per unit area was computed as the product of the average weight per shoot and the number of shoots in the adjacent permanent plot. The below-ground biomass per unit area was calculated similarly from the average rhizome-plus-root weight per shoot. This method of calculation was reported to give reliable results for a *Carex sp.* (sedge) wetland in New York (Bernard and Gorham, 1978).

During the 1996 survey, five representative sites were randomly chosen along each transect for collection of biomass samples. As in the 1991 survey, samples of above-ground tissue were harvested from a 3.3 x 3.3-ft (1-m^2) area adjacent to the transect of permanent plots, but the samples of below-ground tissue were collected within these areas from a larger 1.6 x 1.6-ft (0.5 x 0.5-m) area to a depth of about 12 in. (30 cm). Samples were processed as described for the 1991 survey.

Preparation for chemical analysis. A small subsample of each above-ground sample and each below-ground sample was processed for chemical analysis before drying. These subsamples were washed in a solution of detergent and 0.4-percent hydrochloric acid for 30 s, rinsed twice in distilled water, oven-dried at 70˚C, and ground in a Wiley mill with a 40-mesh screen. The 1991 samples were composited according to soil wetness at the permanent plots, and at least three composite samples from each of three “wetness” environments (dry, shallow-water, and deep-water areas) along each transect were processed. Each of the five 1996 samples was processed individually. All samples were analyzed for metals (copper, nickel, chromium, cobalt, molybdenum, zinc, aluminum, iron, boron, manganese, and lead) and nutrients (nitrogen, phosphorus, potassium, calcium, magnesium, and sodium) at the Soil and Plant Testing Laboratory in the Soils, Crops and Atmospheric Sciences Department at Cornell University, Ithaca, N.Y.

1999 and 2001 Protocols

Sample collection and biomass calculation. Biomass samples were harvested from randomly selected sites between every third and fourth permanent plot along each transect. Above- and below-ground biomass samples were collected from a 0.82 x 0.82-ft (0.25 x 0.25-m) area at each site; below-ground samples were collected to a depth of about 12 in. (30 cm) and rinsed free of sediment and debris in the field. All samples were stored at 4°C to minimize weight loss through respiration before processing. Samples were then heated to 65°C and dried to constant weight; biomass per unit area was computed as the product of mass per cattail shoot and density of cattail shoots.

Preparation for chemical analysis. Cattail tissues were ground and sieved through a 60-mesh screen, and 1-g samples of dried material were ashed in a muffle furnace at 360 to 400° C for 24 to 36 h. After cooling, the samples were removed and weighed for the loss of carbon on ignition. Each sample was dissolved in 5 mL of 6M nitric acid and 5 mL of concentrated hydrogen peroxide, then diluted with 10 mL deionized water. The samples were stored in plastic bottles and refrigerated. Analysis was done by inductively coupled plasma–atomic emission spectroscopy—the 1999 samples at XRAL Laboratories, Toronto, Canada, and the 2001 samples at Copper State Analytical Laboratories, Tucson, Ariz. Sample preparation for nitrogen-content analyses entailed grinding at least 2 g of each sample to pass through a 60-mesh screen. The 1999 samples were analyzed at the Soil and Plant Tissue Testing Laboratory, University of Massachusetts, Amherst, Mass.; the 2001 samples were analyzed at the Agricultural Analytical Services Laboratory, Pennsylvania State University, University Park, Pa. Percent nitrogen was calculated by the Kjeldahl method (Issac and Johnson, 1976).

Fish Surveys

Fish surveys, conducted during 1991-92, 1996, 1999, and 2001, were done by the Department of Environmental and Forest Biology, SUNY College of Environmental Science and Forestry, at Syracuse (Miller and Ringler, 1992; Crabtree and Ringler, 1998; Thompson and others, 2000; Crabtree and Ringler, 2002). The same contractor was chosen for each survey to ensure procedural uniformity. The survey design was based on the hypotheses that the FCS might (1) act as a physical barrier to fish passage and thereby cause a decrease in use of the southern wetland area by nonresident fish species; or (2) cause a change in nutrient and sediment distribution and accumulation rates (an increase in the southern wetland area and a decrease in the northern wetland area) and thereby alter the growth rates of selected fish species or the trophic status of the fish community. The surveys were designed to identify fish species and to document abundance, diversity, and the distribution of species within the wetland.

Sampling Methods

Three fish-sampling methods were used during most of the surveys—South Dakota trapnets (1.5-m x 1.5-m opening with 1-cm mesh), Gee wire minnow traps (44-cm long x 9-cm diameter with 0.5-cm mesh), and a gill net (50-m x 2-m with 5 panels of mesh sizes 1.2, 2.5, 3.5, 5.1, and 6.3 cm) (table 2). During the 2001 survey, only trapnets were used, and during September 1991, when low water prohibited the use of trapnets, seining and gill nets were used. Other techniques—electrofishing in 1996 and video monitoring of fish passage through the FCS in 2001—were used to test their utility in identifying any effect the FCS might have on the fish community. The same sampling effort was expended upstream and downstream from the FCS during each survey. Generally, one gill net, two trapnets, and 20 minnow traps were deployed in each area for 1, 4, and 6 consecutive nights, respectively, during each of three seasons (late spring, midsummer, and early fall) during a given survey year (fig. 7). Captured fish were identified and tallied; a random subset of black crappie, bluegill, largemouth bass, northern pike, and pumpkinseed were measured for total length and weighed, and a scale sample was collected for calculation of growth rates for these species.

Species Classification

Fish were classified as resident species (those likely to spend all of their lives within a marsh setting) or nonresident species (those that migrate into a marsh environment for only part of their life cycle). This classification was based on a correspondence analysis by Jude and Pappas (1992), which separated species along an environmental gradient or according to habitat type. Any species that was recognized by Jude and Pappas (1992) as a marsh species was classified as a resident species of the wetland; all others were classified as nonresident species.

Age and Growth-Rate Calculation

The age of five species—black crappie, bluegill, largemouth bass, northern pike, and pumpkinseed—was estimated from scale samples. Measurements were made from the center of the scale (focus) to the margin and to every apparent true annulus; the sum of the annuli corresponded to the age of the fish. The length of an individual at an estimated age was adjusted for the growth of the fish before scale formation by the Fraser-Lee method (Bagenal and Tesch, 1978) to obtain the mean length for each age (cohort) within a species. This calculation was based on the total length of the fish, the radius of the scale, and the radius of an individual annulus. The mass of an individual at an estimated age was calculated from the age of the fish and the measured mass or, if only length was measured, from a linear regression of mass and length. The mean mass was the average mass of all fish of a specific age. The instantaneous growth rate for a species was calculated as

Table 2. Sampling methods for fish surveys in Ellison Park wetland, Monroe County, N.Y., 1991-2001

[Number, number of devices or techniques applied. DS (US), downstream (upstream) from flow-control structure. Sampling days, number of sampling days during given survey (generally divided equally among three sampling periods each year).]

Sampling method	1991-92 Number DS	1991-92 Number US	1991-92 Sampling days 7/91, 5/92	1991-92 Sampling days 9/91[a]	1996 Number DS	1996 Number US	1996 Sampling days	1999 Number DS	1999 Number US	1999 Sampling days	2001 Number DS	2001 Number US	2001 Sampling days
South Dakota trap nets	2	2	8	0	2	2	12	2	2	12	2	2[b]	12
Gee minnow traps	20	20	12	6	20	20	18	20	20	18	0	0	0
Gill nets	1	1	2	6	1	1	3	1	1	2	0	0	0
20-foot bag seine hauls	7	0	0	1	0	0	0	0	0	0	0	0	0
Electrofishing sites	0	0	0	0	9	9	1	0	0	0	0	0	0
Video monitoring	-	-	-	-	-	-	-	-	-	-	-	-	3

[a] Extremely low water levels during September 1991 made seining and increased use of gill nets necessary.

[b] Extremely low water levels prevented use of trap nets in wetland area upstream from flow-control structure.

the difference between the mean mass of a cohort for year (n) and that of a cohort for year (n-1) for each species (Bagenal and Tesch, 1978).

Trophic Classification

Fish species were classified as insectivore, invertivore, omnivore, or piscivore on the basis of Plafkin and others (1989). Percentages of species in each trophic group were computed for each season and for each wetland area (upstream and downstream from the flow-control structure).

Contaminant-Related Stress Indicators

Brown bullhead have been used as biological indicators of stress associated with contaminated sediments (Hickey and others, 1990; Hirethota and Ringler, 1993). Sediment from highly urbanized areas can contain high concentrations of potentially toxic chemicals, such as metals and organic compounds, and can accumulate in depositional areas such as wetlands and pose a hazard to bottom-feeding fish. Brown bullhead were examined for signs of contaminant-related stress, including barbel deformities, melanotic pigmentation, lesions, ulcers, and tumors.

Benthic-Macroinvertebrate Surveys

Benthic-macroinvertebrate surveys were conducted during 1991-92 by the Department of Environmental and Forest Biology, SUNY College of Environmental Science and Forestry, at Syracuse (Miller and Ringler, 1992), and during 1996 by the Department of Biological Sciences, SUNY College at Brockport (Haynes and others, 1998). No survey was conducted after installation of the FCS.

The design of the 1991-92 survey was to provide a "snapshot" of the benthic-macroinvertebrate community through identification and enumeration of individuals found in the stomach contents of fish captured during July or September 1991 or May 1992 (Miller and Ringler, 1992). Stomachs were removed from formalin-preserved fish, and the contents were flushed, identified, and enumerated. Insects were keyed to the lowest possible taxonomic unit, which generally was family.

The design of the 1996 survey was more comprehensive than the 1991-92 survey and, in anticipation of similar surveys after FCS installation, was based on the hypothesis that the FCS might cause a change in the distribution and accumulation rates of sediment and sediment-associated pollutants (an increase in the southern wetland area and a decrease in the northern wetland area) and thereby alter the composition and species abundance of the benthic-macroinvertebrate community. The survey was designed to identify benthic-macroinvertebrate species and to document abundant, present, and absent species and the distribution of species within the wetland. A survey of the macroinvertebrate communities was conducted from late September through October 1996 at creek-channel and backwater-channel (wetland) sampling sites in the southern part of the wetland, and at two similar sites in the northern part. Three microhabitats—channel bottom, vegetation, and pelagic (open-water) zone—were sampled at each location (Haynes and others, 1998). The channel bottom at the creek site in the southern part of the wetland was mostly sand and was sampled by the kick method (Bode and others, 1990); the channel bottoms of the other three sites were muddy and were sampled with an Ekman dredge. Submergent and emergent vegetation were sampled with D-ring dip nets. The open-water areas at these sites were sampled by multiplate samplers (Bode and other, 1990), which were deployed during a 5-week period beginning September 25.

Five replicate samples were collected in each of the three microhabitats at each site. Samples were preserved in the field in 4-percent formalin solution. After 24 hours, samples were transferred to 70-percent ethanol containing rose bengal dye. Macroinvertebrates were separated from debris and identified to the lowest practicable taxonomic unit. Chironomid mouthpart deformities were identified through criteria of Warwick (1990, 1989). Physical characteristics of each sampling site, including water temperature, specific conductance, dissolved-oxygen concentration, pH, flow velocity, and water depth, were measured, as were percent canopy cover, amount of vegetation, and substrate composition, to define the comparability among data collected at the two creek sites and the two wetland sites. Community and species indicators of water quality were identified, and water-quality indices were calculated.

Bluegill (*Lepomis macrochirus*) and pumpkinseed (*L. gibbosus*) were collected by electrofishing. Stomach contents were removed from 20 fish of each species and preserved in formalin. The macroinvertebrates contained therein were identified and compared with those identified from the stomach contents of fish captured during 1991-92.

Bird Surveys

The New York State Department of Environmental Conservation and the Federation of New York State Bird Clubs had conducted a statewide survey during 1980-85 that included the Ellison Park wetland to identify bird species and map the extent of their breeding ranges (Andrle and Carroll, 1988). Birds were identified by sight, sound, and nest or egg identification and were classified as possible, probable, or confirmed breeders on the basis of breeding evidence. *Possible* breeders were those species observed in breeding season in possible nesting habitat or whose breeding calls were heard. *Probable* breeders were those species that exhibited breeding activities, including breeding calls heard on more than one date in the same place, territorial behavior, courtship displays, visitation of probable nesting site, and nest building or excavation of a nest hole. *Confirmed* breeders were those species for which definite evidence of breeding was found, such as a used nest, an identifiable nest with eggs, fledglings, and adult carrying food for young.

The design of the bird surveys that were conducted during the Ellison Park wetland study (1990-2001) was based on the hypotheses that the FCS might (1) cause changes in the timing, duration, and depth of high-water levels during the breeding seasons of ground-nesting birds, and (2) increase sedimentation rates that would prematurely fill open-water areas. The surveys were designed to identify species and document their presence or absence. Two precontrol bird surveys and one postcontrol survey were conducted. Miller and Ringler (1992) identified bird species by sight or call on July 9-15 and September 23-29, 1991, and May 19-21 and 28-30, 1992, while canoeing in the wetland for the 1991-92 fish survey, but did not identify the breeding status of these species. McKinney (1996) conducted a survey of the breeding birds of the wetland by walking the wetland perimeter and making nine canoe trips into the wetland during June 10 to July 22, 1996. Spahn (2002) visited the wetland five times during May 17 to July 11, 2002—four trips on foot and one by canoe. The 1996 and 2002 surveys used bird calls and recordings of bird songs to elicit responses, and documented evidence of breeding by the same criteria as used by Andrle and Carroll (1988).

Sediment Studies

Two main ecological concerns regarding installation and operation of the FCS were (1) the probable increase in the deposition of fine-grained sediment (silt and clay) in the southern wetland area and the accelerated filling-in of open-water areas in the wetland, and (2) an increase in the concentration of metals and organic compounds that are associated with fine-grained sediments that could adversely affect the wetland biota, including benthic macroinvertebrates, amphibians, reptiles, bottom-feeding fish, and the predators of these animals. To address these concerns, the USGS made annual measurements of sediment accumulation (1992-2002) and periodically collected sediment samples (during 1994, 1997, 1999, and 2001) for chemical analyses.

Sedimentation Rate

Measurement of sedimentation rates entailed anchoring 1-ft-diameter plastic disks to the cattail mat in depositional environments and adjacent to open-water areas of the wetland (fig. 8) to provide reference surfaces from which sediment depths could be measured. Measurements were made annually at 6 or 7 points on each disk; the average of these values was used as the sediment thickness. Annual inspections coincided with seasonally low water levels from October through November to ensure that the disks were not submerged at the time of inspection. Measurements were adjusted for thickness of any organic matter that had accumulated on the disk, primarily duckweed and decaying cattails. Sedimentation rates (mm/yr) were computed as the net sediment thickness divided by the durations of the precontrol and postcontrol periods (in years). Use of a length measurement (mm) rather than a weight measurement by volume (g/cm^3) or area (g/m^2) permitted comparison of current sedimentation rates with previously published historical and near-recent rates. Alternative methods were considered, including surveying along permanent transects (Ritchie and McHenry, 1985), using sediment traps similar to those used in lake-sedimentation studies, using radioactive-isotopic dating of sediment cores (Ritchie and McHenry, 1990), and using a dendrogeomorphic technique to relate tree age to sediment depth over the tree's root collar and initial-growth root zone (Hupp and Morris, 1990), but these methods were not used because (1) the surveying technique is imprecise in wetland environments, (2) sediment traps tend to measure total sediment influx rather than net deposition or a sedimentation rate that represents the natural deposition and removal processes in a wetland, and (3) radio-isotopic coring and dendrogeomorphic measurements provide only long-term sedimentation rates.

Sediment Quality

Samples of fine-grained sediment were collected at three sampling sites in the Ellison Park wetland during the fall of 1994, 1997, 1999, and 2001 (fig. 8). Two of the sites were upstream from the FCS, and one was downstream; all were within open-water, long-term depositional areas and distant from channels that might carry high-velocity, erosive flows. Samples were carefully scooped under shallow water from the top 2 to 3 in. of sediment, packed in ice, and shipped to the USGS National Water Quality Laboratory in Arvada, Colo., by methods described by Shelton and Capel (1994) and Radtke (1997). No samples contained particle sizes greater than sand (2 mm and greater). All of each sample was analyzed for (1) total organic carbon, (2) polycyclic aromatic hydrocarbons by gas chromatography with a mass-spectrometric detection, and (3) organochlorine compounds by gas chromatography with electron-capture detection as described by Wershaw and others (1987); and (4) organophosphate compounds by gas chromatography with flame-photometric detection as described by Jha and Wydoski (2003). The clay fraction (particle size < 63 μm) was analyzed for major and trace elements by procedures listed by Timme (1995) and described by Fishman and Friedman (1989). Separate samples were collected and submitted to the USGS Sediment Laboratory in Iowa City, Iowa for particle-size analyses. Concentrations of selected constituents were compared with those published as sediment-quality guidelines for assessment of the probability of adverse effects to aquatic life.

Base from New York State Plane NAD 83 Orthoimagery at URL
http://www.nysgis.state.ny.us/gateway/mg/monroe_download.htm, 2002

Figure 8. Locations of data-collection sites for sedimentation-rate and sediment-quality studies in Ellison Park wetland, Monroe County, N.Y., 1991-2002

Surface-Water Data

Stream discharge and water-quality were monitored at the USGS streamflow-monitoring stations at the inlet (Blossom Road) and outlet (Empire Boulevard) of the wetland (fig. 2) throughout the study period. Each monitoring station included an electronic data logger, water-stage and temperature sensors, and an automatic water sampler. Sensors made measurements every 15 min, and the measured values were stored in the data loggers.

Stream Discharge

Discharge at the upstream site was computed from a stage-to-discharge relation by standard USGS procedures for gaging streamflow (Carter and Davidian, 1968; Rantz and others, 1982), measuring stage and discharge (Buchanan and Somers, 1968 and 1969, respectively), and analyzing stage-to-discharge relations (Kennedy, 1984). Computation of discharge at the downstream site was complicated by backwater conditions caused by fluctuating water levels in Irondequoit Bay and Lake Ontario; therefore, water velocity was measured with an acoustic velocity meter (AVM), and the recorded value was correlated with the mean velocity in the channel as calculated from discharge measurements made by current meter from the Empire Boulevard bridge or by acoustic Doppler current profiler (ADCP) from a boat. Discharge was computed from the velocity relation and from the relation between stage and flow area. Standard USGS procedures for measuring discharge with an ADCP (Lipscomb, 1995), and gaging streamflow with an AVM and analyzing stage-and-velocity-to-discharge relations (Laenen, 1985) were followed.

Water Quality

Automated samplers that extracted water samples from the channel (near the centroid of flow) hourly and stored them in refrigerated bottles were maintained by the Monroe County Environmental Health Laboratory (MCEHL). Samples were retrieved twice weekly and delivered to MCEHL. Sampling periods were divided on the basis of three flow conditions—base or steady flow, and the rising and falling phases of a storm hydrograph. If base- or steady-flow conditions prevailed during the entire sampling cycle (3 to 4 days), equal volumes of water from all samples collected during that cycle were composited for a single analysis. During storms, samples collected during the rising phase were composited separately from those collected during the falling phase. Equal volumes of water from all samples collected during a given phase were composited for a single analysis. Laboratory analyses were done by MCEHL, which participated in the USGS quality-assurance program for cooperating analytical laboratories. Analytical procedures are described in American Public Health Association and others (1995). Samples were analyzed for phosphorus (total and orthophosphate), nitrogen (total, ammonia-plus-organic, nitrate-plus-nitrite, and ammonia), chloride, sulfate, and suspended solids (total and volatile). Samples collected during stormflows, and periodically during low-flow periods, were also analyzed for total organic carbon, biochemical and chemical oxygen demand, alkalinity, specific conductance, and trace metals (zinc, lead, copper, and cadmium).

Quality Assurance and Quality Control

Quality-assurance and quality-control protocol (QA/QC) for the water-sampling and analysis program conducted by MCEHL included comparison of constituent concentrations measured in depth-integrated cross-sectional samples collected by the equal-width-increment (EWI) method (Shelton, 1994) with those measured concurrently in samples collected with automatic samplers (Auto). Statistical analyses (paired t-test; $\alpha = 0.05$) of the mean of the differences between paired data—(EWI and Auto)—identified any significant systematic bias in the concentrations in the automatically collected water samples.

MCEHL also participated in the semiannual USGS Standard Reference Sample (SRS) program for cooperating analytical laboratories, which is coordinated by the USGS Office of Water Quality. Participating laboratories receive (1) reference samples, which are analyzed for trace elements, major ions, and nutrients; and (2) low-ionic strength samples that simulate precipitation. The analytical results from all participating laboratories are transmitted to USGS, and a "most probable value" (MPV) is statistically calculated for each constituent. MCEHL results were consistently rated "satisfactory" to "good" (within 1.50 to 0.51 standard deviations of the MPV) for trace elements; and "good" to "excellent" (within 1.00 to 0.00 standard deviations of the MPV) for major ions, nutrients, and mercury, and for low-ionic-strength samples (table 3).

Constituent Loads and Removal Efficiency

Monthly and annual loads of phosphorus compounds, nitrogen compounds, suspended solids, chloride, and sulfate at the two water-quality-monitoring stations were calculated by the USGS program ESTIMATOR (Gregory Baier, Timothy Cohn, and Edward Gilroy, U.S. Geological Survey, written commun., 1995), which is based on a log-linear regression equation that relates nutrient concentrations to surrogate variables of discharge, time (year and decimal-date), and season (Cohn and others, 1989; 1992). Nine variables were considered for each equation: a constant; a quadratic fit to the logarithm of discharge (log flow and square of log flow); a quadratic fit to time (decimal time and square of decimal time); and two sinusoidal (first- and second-order Fourier) functions to account for the effects of annual seasonality (Cohn and others, 1992).

Pre- and postcontrol loads were computed from data collected during each respective period. The loads for October

Table 3. Performance ratings for selected standard reference samples analyzed by Monroe County Environmental Health Laboratory, 1990-2001 (results of U.S. Geological Survey's analytical evaluation program)

[Rating scale: 4, excellent; 3, good; 2, satisfactory; 1, marginal; 0, unsatisfactory. *n*, number of samples analyzed. bdl, concentration below detection limit of analytical equipment used by laboratory.]

Date of analyses	Overall weighted rating	*n*	Nutrients[a]							Major ions[b] and physical properties								Trace metals[c]					
			Rating[d]	*n*	NH_3	NH_3+ Org N	NOx	TP	PO_4	Rating[4]	*n*	Alk	Cl	SO_4	TP	pH	SC	Rating[4]	*n*	Cd	Cu	Pb	Zn
May 1990	3.3	51	3.3	11	3	3.5	2	4	4	3.6	13	4	3	3	4	4	3	3.1	15			4	4
Nov. 1990	3.1	59	3.2	22	2	3.5	4	4	3.5	3.5	12		4	4	3	3	4	2.5	15	3	4		4
Apr. 1991	3.0	56	3.6	14	4	3.8	4	3.8	4	3.0	13	4	4	3	4	4	0	2.3	18	3	4	4	3
Oct. 1992	3.1	55	3.7	18	3.8	3	4	3.8	4	3.7	13	4	4	4	4	4	3	2.0	15	3	bdl	2	bdl
Apr. 1993	3.0	59	3.8	18	4	3.2	4	3.8	3.8	2.7	13	3	0	0	4	4	4	2.6	28	3	bdl	3.5	bdl
Oct. 1993	3.0	97	3.8	-	4	3.5	4	3.8	3.5	3.5	13	4	4	4	4	4	3	2.5	18	3	2	4	3
Oct. 1994	2.6	75	3.7	10	3	4	4	4	3.5	3.2	13	4	4	4	3	2	2	2.2	40	0.5	2	4	2
May 1995	2.9	57	3.3	10	3.5	3	3.5	3.5	3	3.4	13	4	4	4	4	0	3	2.5	22	3	2	4	0
Oct. 1995	3.1	54	3.4	10	3.5	2	4	3.5	4	3.2	13	4	3	4	2	4	3	2.7	19	2	bdl	2	1
Apr. 1996	2.6	68	3.8	5	4	4	3	4	4	3.2	13	4	1	0	4	4	3	2.3	38	3.5	2	0	1.5
Sept. 1996	3.0	76	3.4	9	3.5	2.5	4	4	3.5	3.0	13	4	3	4	4	3	3	2.8	43	3.5	3.5	3.5	0.5
Apr. 1997	2.9	85	3.6	10	3.5	3.5	3	4	4	3.2	14	4	3	4	4	4	4	2.5	35	3	bdl	0	4
Sept. 1997	2.7	102	3.7	10	3.5	3.5	3.5	4	4	3.2	13	3	4	3	3	4	4	1.7	17	0	3	0	0
Apr. 1998	2.7	78	3.8	10	4	4	3.5	3.5	4	2.6	13	3	4	3	bdl	4	3	2.4	20	4	3	4	1
Sept. 1998	2.6	72	3.6	10	3.5	3.5	3.5	3.5	4	2.8	11	bdl	4	4	3	4	2	2.4	20	0	2	2	3
Mar. 1999	2.9	76	3.1	8	4	4	3	3.5	1.5	2.9	13	3	4	3	0	4	2	2.3	22	1	2	2	3
Oct. 1999	2.5	57	3.5	10	3.5	3.5	3.5	4	3	2.2	13	0	3	4	4	0	4	2.1	22	2	3	3	3
Mar. 2000	2.9	59	3.7	10	3.5	3.5	3.5	4	4	3.1	14	4	4	4	4	2	3	2.3	23	4	0	3	1
Oct. 2000	2.7	59	3.7	10	4	4.5	4.5	3.5	4	2.7	14	4	4	3	4	4	0	2.0	23	2	3	0	4
Apr. 2001	2.8	59	3.8	10	3.5	4	3.5	4	4	3.3	14	4	4	4	4	4	4	1.7	23	0	bdl	1	2
Sept. 2001	2.6	57	3.5	10	3.5	3.5	3	3.5	4	3.1	14	4	4	4	4	4	4	2.1	22	0	4	2	2
Average	2.9		3.6		3.6	3.5	3.6	3.8	3.7	3.1		3.8	3.4	3.3	3.7	3.3	2.9	2.3		2.2	2.6	2.4	2.2

[a] NH_3, ammonia; NH_3+Org N, ammonia-plus-organic nitrogen; NOx, nitrate plus nitrite; TP, total phosphorus; PO_4, orthophosphate.

[b] Alk, alkalinity; Cl, chloride; SO_4, sulfate; SC, specific conductance.

[c] Cd, cadmium; Cu, copper; Pb, lead; Zn, zinc.

[d] Overall rating for constituent category.

Table 4. Explanatory variables used in regression equations developed to compute loads for selected constituents in water from Irondequoit Creek, Monroe County, N.Y., 1990-2001, at sites above Blossom Road (inflow to wetland) and Empire Boulevard (outflow from wetland)

[T, time, in years. Coefficient for specified variable was found to be significantly different from zero (p-value less than 0.05) for one or both of the inflow-load and outflow-load equations. If, for a given constituent, a variable was significant for only one of the equations, it was nevertheless used in both equations to maintain comparability of results.]

Constituent	Constant	Log flow	Log flow squared	Decimal time	Decimal time squared	Seasonality function			
						Sin(2πT)	Cos(2πT)	Sin(4πT)	Cos(4πT)
Total phosphorus, as P	X	X	X	X	X	X	X	-	-
Orthophosphate, as P	X	X	X	X	X	X	X	X	X
Ammonia plus organic nitrogen, as N	X	X	-	X	-	X	X	X	X
Ammonia nitrogen, as N	X	X	-	X	-	X	X	X	X
Nitrate plus nitrite, as N	X	X	X	X	X	X	X	X	X
Total suspended solids	X	X	-	X	X	X	X	-	-
Volatile suspended solids	X	X	-	X	X	X	X	-	-
Chloride	X	X	X	X	-	X	X	X	X
Sulfate	X	X	X	X	X	-	-	-	-

1990 through September 1996 were those computed by Coon (2000). The loads for October 1996 through February 1997, when the base control was installed, were computed from the data collected from October 1990 through February 1997, and were appended to the loads calculated by Coon (2000) to complete the precontrol data set. Postcontrol loads were calculated from data collected from March 1997 through September 2001.

Each dataset was used to develop a load equation for each constituent at each sampling site. For a given constituent, if the coefficient for a variable was significantly different from zero ($p < 0.05$) for either the inflow- or outflow-load equation, that variable was included in both equations to maintain comparability of the results (table 4). The postcontrol equations used the same variables as the precontrol equations. The constituent loads were computed from continuous water-quality data (composited hourly samples); typically at least two samples were provided per week, but more were obtained during storms, when samples were composited during the rising and falling phases. The discharge associated with a given water sample was computed as the mean of the 15-min discharges indicated by the stage-to-discharge relation at the Blossom Road monitoring station and the stage-and-velocity-to-discharge relation at the Empire Boulevard site for the period covered by the hourly water samples that were composited into the analyzed sample.

Daily loads were computed by the load equation from the daily mean discharges recorded at each site. From these data, total monthly, annual, and period-of-record loads were computed for each constituent. (See appendix.) The precision of the estimated loads can be described in terms of a confidence interval, which is based on the estimated mean and the standard error of prediction calculated by the equation. At a 95-percent confidence interval ($\alpha = 0.05$), the confidence limits are the estimated load ±1.96 x the standard error of prediction. (See appendix.)

Potential problems in calculating loads from regression equations include (1) multiply censored water-quality data (which concentrations are found to be below an analytical detection limit that can change with improvements in analytical measurement capabilities), (2) logarithm-retransformation bias, and (3) lack of fit between predicted and observed values. The ESTIMATOR program handles multiply censored data through an adjusted maximum-likelihood estimation procedure (Cohn, 1988; 1995). Loads computed from the logarithm of constituent concentrations can show bias when the loads are transformed back into nonlogarithmic units. The ESTIMATOR program corrects this bias, which tends to underestimate the loads, through a minimum-variance unbiased estimator (Cohn, 1988; 1995; Cohn and others, 1992). Misspecification of a regression equation, which is likely to result when linear regression is used to describe the relation among time-series data, can be indicated by serial correlation (nonindependence) of the residuals. The ESTIMATOR program minimizes serial correlation of residuals and the effect of seasonality by including sinusoidal functions of time ($\sin 2\pi T$, $\cos 2\pi T$, $\sin 4\pi T$, $\cos 4\pi T$, where T is time, in years) (Cohn and others, 1992). Lack of fit of a regression equation can be indicated by a low coefficient of determination and by nonnormal distribution and serial correlation of the residuals. Cohn and others (1992) point out that (1) load estimators based on log-linear equations, in general, appear to be relatively insensitive to violations of the assumptions of linear regression—that is, nonnormality and independence of the residuals; (2) a minimum variance unbiased estimator, in particular, which is based on the assumptions of a log-linear equation, provides reliable estimates of loads despite these violations; and (3) the variances of annual or monthly load estimates based on infrequent sampling appear to be well described by log-linear equation theory. Equation validity or goodness of fit was evaluated through regression diagnostic statistics—standard error of prediction (in appendix), coefficient of determination, standard error of regression, serial correlation of residuals, and probability plot-correlation coefficient (a measure of the normality of the residuals)—with consideration given to the number of total and censored observations included in the development of each equation (table 5).

The USGS and MCEHL collected an extensive streamflow- and water-quality data set. As a result, the chemical loads could have been computed directly from the concentration data by means of a spreadsheet, but the ESTIMATOR program was chosen because it provided easier, more efficient computation. The validity of using the ESTIMATOR-computed loads rather than manually computed loads was tested through a statistical comparison of manually computed monthly 1997-98 loads with the ESTIMATOR loads. The Wilcoxon signed-ranks test in SAS (SAS Institute, Inc., 1989; 1990) was used to identify statistically significant differences between the values obtained by each method. Of 20 comparisons of 1997-98 loads of phosphorus compounds, nitrogen compounds, suspended solids, chloride, and sulfate at the inflow and outflow sites, 17 indicated no significant differences ($p = 0.1180$ to 0.9779) between the two computation methods; the remaining three indicated significant differences for ammonia, ammonia-plus-organic nitrogen, and total nitrogen loads at the outflow site only; the corresponding p values were 0.0010, 0.0064, and 0.0396. These results indicated that use of the ESTIMATOR program to compute loads was a valid alternative to manual computation of loads.

The wetland's effectiveness in decreasing chemical loads can be evaluated in terms of removal efficiency—the percentage of an inflow load that is retained in, or exported from, the wetland. Removal efficiency for a given constituent was computed as the difference between inflow and outflow load divided by the inflow load and expressed as a percentage. Monthly and annual removal efficiencies were computed for each constituent. (See appendix.) A positive value indicated a net retention of a constituent in the wetland, and a negative value indicated the generation of a constituent within the wetland (or, possibly, its adjacent drainage area) and a net export of a constituent.

Statistical Analyses of Load Data

The computed loads were statistically analyzed through SAS (SAS Institute, Inc., 1989; 1990) to provide an objective basis for assessing the effectiveness of the FCS in decreasing chemical loads. Nonparametric tests were used because most of the monthly load data did not show normal distributions. Three approaches were followed:

(1) Statistically significant differences between the monthly inflow and outflow loads were identified by the Wilcoxon signed-ranks test on the ranked differences between the paired (inflow and outflow) loads. This test indicated whether the median of the differences between inflow and outflow loads was significantly different from zero.

(2) Three major high flows during the postcontrol period exceeded all of the peak flows that occurred during the precontrol period. This dissimilarity could greatly affect the interpretation of the FCS's ability to decrease chemical loads; therefore, a second set of data was generated. The estimated monthly loads (in tons) were divided by the total monthly volume of water (in ft^3) passing the respective streamflow-monitoring sites, and the resulting values were converted to monthly flow-weighted concentrations (in mg/L). Wilcoxon signed-ranks tests were performed on the ranked differences between the paired (inflow and outflow) flow-weighted concentrations.

(3) The precontrol data reflected the effect of the wetland on chemical loads, whereas the postcontrol data reflected the combined effects of the wetland and the FCS. The wetland's effect was removed from a given analysis by adding or subtracting, as necessary, the mean precontrol difference—inflow load or flow-weighted concentration for a given constituent minus its outflow load or flow-weighted concentration, as described above—from the differences between the postcontrol inflow and outflow loads or flow-weighted concentrations. Theoretically, the result would represent the change in chemical load that could be attributed solely to the FCS. These adjustments were made to the load- and the flow-weighted-concentration differences, and the Wilcoxon signed-ranks tests were again performed.

EFFECTS OF FLOW MODIFICATION

Throughout the following discussion of the effects of flow modifications on the Ellison Park wetland, references are made to the precontrol period (or before FCS installation) and the postcontrol period (or after FCS installation). In all instances, these references are meant to include not only the installation of the FCS, but also the changes in flow patterns that resulted from the bank notches and diversions from Irondequoit Creek to the Millrace upstream from the wetland, as described in the section "Additional Flow Modifications" (p. 11). Although these additional flow modifications were made over a period from February 1997 through the summer of 1998, the pre- and postcontrol periods have been defined by the month of the FCS installation (February 1997) for comparison purposes.

The FCS was expected to have several measurable effects. First, it was designed to attenuate the discharge of many, if not all, peak flows passing through the wetland and to cause some amount of backwater upstream. It also was expected to increase the dispersal and detention of stormflows in the upstream part of the wetland and thereby affect the loads of chemical constituents carried by Irondequoit Creek into Irondequoit Bay. The FCS, as well as the other flow modifications, could increase the sedimentation rate of fine-grained sediments, which in turn

Table 5. Regression statistics for equations used to compute constituent loads for Irondequoit Creek sampling sites above Blossom Road and at Empire Boulevard, Monroe County, N.Y., 1990-2001

Constituent	Precontrol period (October 1990 through February 1997)						Postcontrol period (March 1997 through September 2001)					
	Number of observations		Coefficient of determination[a]	Standard error of regression[b]	Serial correlation of residuals[c]	PPCC[d]	Number of observations		Coefficient of determination	Standard error of regression	Serial correlation of residuals	PPCC
	Total	Censored					Total	Censored				
Blossom Road												
Total phosphorus, as P	1080	0	0.748	0.664	0.314	0.988	603	0	0.812	0.587	0.219	0.979
Orthophosphate, as P	1081	3	.764	.477	.508	.989	631	2	.891	.345	.644	.986
Ammonia-plus-organic nitrogen, as N	1075	2	.851	.381	.382	.989	601	5	.860	.434	.332	.993
Ammonia nitrogen, as N	1044	473	.574	.855	.432	.953	625	338	.709	.899	.346	.955
Nitrate plus nitrite, as N	1065	0	.939	.214	.635	.950	591	0	.960	.171	.552	.984
Total suspended solids	324	4	.664	.838	.197	.982	191	0	.708	.944	.030	.991
Volatile suspended solids	320	23	.647	.724	.271	.986	189	33	.704	.841	.056	.988
Chloride	1086	0	.928	.196	.578	.974	595	0	.937	.208	.547	.971
Sulfate	1074	1	.853	.174	.177	.936	595	0	.891	.155	.129	.962
Empire Boulevard												
Total phosphorus, as P	940	0	.836	.445	.404	.977	572	0	.843	.448	.445	.980
Orthophosphate, as P	941	1	.871	.320	.535	.983	670	1	.903	.330	.637	.972
Ammonia-plus-organic nitrogen, as N	928	0	.876	.309	.468	.981	574	5	.872	.376	.268	.961
Ammonia nitrogen, as N	911	99	.523	.700	.575	.986	660	29	.772	.528	.591	.990
Nitrate plus nitrite, as N	929	0	.958	.186	.624	.991	599	0	.970	.159	.625	.994
Total suspended solids	243	3	.733	.633	.224	.984	159	1	.691	.805	.156	.989
Volatile suspended solids	242	29	.752	.504	.292	.987	159	27	.681	.734	.253	.987
Chloride	944	0	.930	.200	.572	.978	602	0	.941	.202	.641	.976
Sulfate	928	0	.874	.163	.246	.975	600	0	.908	.145	.231	.983

[a] Coefficient of determination (R^2) - the percentage of the total variance in the computed loads that is accounted for by the regression equation. Values close to 1.00 are desired.

[b] Standard error of regression - a measure of the dispersion of the data around the regression line. Low values are desired.

[c] Serial correlation of residuals - a measure of the dependence or correlation in time sequence between residuals. The serial correlation coefficient ranges from zero (no serial correlation) to 1.00 (strong serial correlation).

[d] Probability plot correlation coefficient - a measure of the likelihood that the residuals are normally distributed. A value greater than 0.97 implies that the data are probably from a population with a normal distribution.

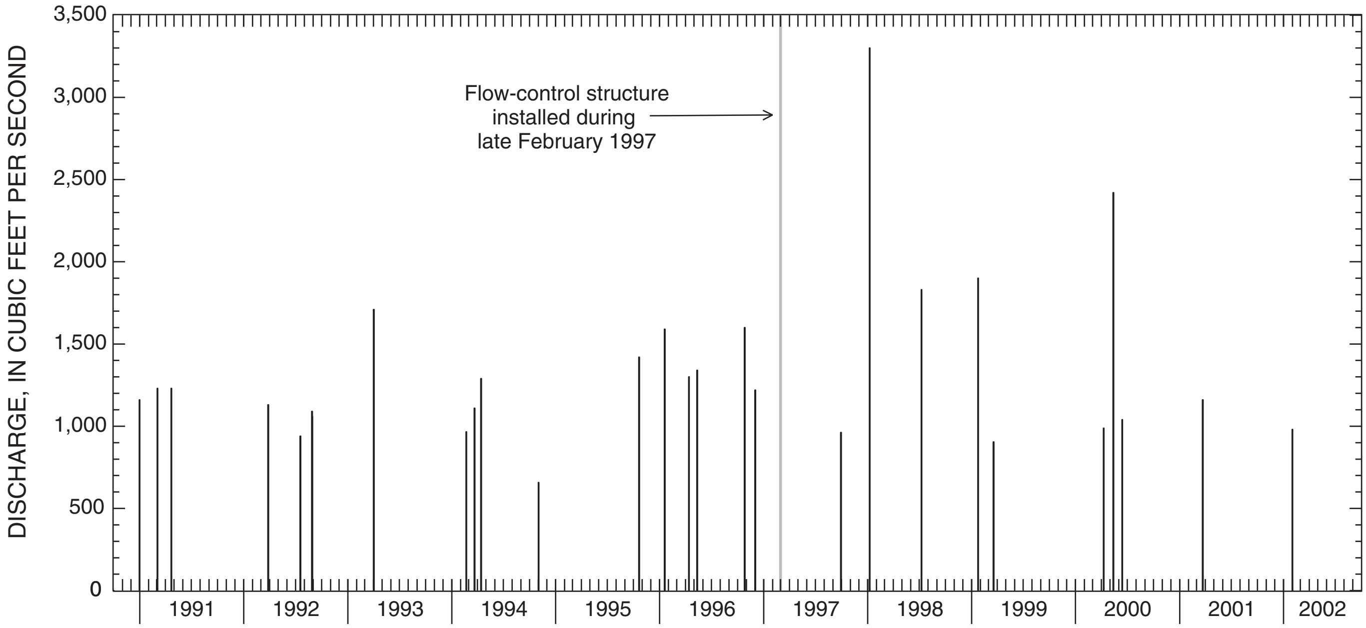


Figure 9. Peak discharges recorded at Irondequoit Creek above Blossom Road, Rochester, N.Y., 1990-2001

could (1) accelerate the infilling of ponded areas in the upstream part of the wetland, and (2) increase the concentrations of sediment-associated chemicals in depositional sites in that area. The flow modifications also could cause changes in the wetland plant, fish, benthic-macroinvertebrate, and bird communities.

The interpretation of results took into account the effects of precipitation and streamflow differences between the pre- and postcontrol periods, as follows. Precipitation quantities and monthly mean streamflows (fig. 3) were more extreme during the postcontrol period than during the precontrol period. Monthly precipitation for July 1992, October 1995, and June 1996 (the precontrol period) greatly exceeded the long-term average, whereas two prolonged wet periods occurred during the postcontrol period June through August 1998 and April through September 2000. One- to 3-month periods of below-average monthly precipitation occurred throughout the study prolonged dry periods of at least 4 months occurred from May through August of 1991 and 1993 (4 months), and from January through June of 1995 (6 months) of the precontrol period. Dry periods occurred from April through July of 1997 and 2001 (4 months), and from September 1998 through July 1999 (11 months) of the postcontrol period.

Peak discharges, which included annual peaks as well as flows that exceeded a base discharge of 900 ft^3/s, were much lower during the precontrol period than during the postcontrol period (fig. 9). The maximum flow during the period of record at the Blossom Road streamflow-monitoring station before installation of the FCS (1982 through February 1997) was 1,730 ft^3/s. Of the nine peak flows that occurred during the 4.5 years after the FCS installation, four exceeded the precontrol maximum flow, and one of these, which occurred within a year of the FCS installation, was the peak of record (3,300 ft^3/s on January 8, 1998—almost twice the previous maximum peak). These peak flows greatly affected the chemical loads that entered the wetland during 1998 and 2000.

The seasonal periods of wetland inundation during the precontrol period were comparable to those of the postcontrol period, in that both periods contained persistently inundated (wet) and noninundated (dry) growing seasons (fig. 4). The prolonged wet condition of 1992-93 did not recur during the postcontrol period, but 1997 and 1998 had two successive wet periods that, at least during the growing season, were comparable to the 1992-93 wet period. Dry, low-water years occurred during each period; 1995 during the precontrol period, and 1999, which was the drier of the two, during the postcontrol period.

Water-Surface Elevations and Discharge

Water-surface elevations during the postcontrol period were compared with precontrol water-surface profiles and those predicted by the WSPRO model. Measurements were made during five storms with peak flows that ranged from 962 to 3,300 ft^3/s, the latter of which was the period-of-record (21 years) peak flow at the upstream (Blossom Road) streamflow-monitoring site (table 6). Flooding in Ellison Park occurs when discharge exceeds about 900 ft^3/s. The measured water-surface elevations were lower and the measured falls in water-surface elevations from Blossom Road to the Narrows were greater than those predicted by the WSPRO model. These results reflect the conservative assumptions and parameter values used in the model.

The discharge measurements made at Blossom Road during the five storms were compared with the established stage-to-discharge relation at the streamflow-monitoring station above Blossom Road. Measured discharges ranged from 311 to 2,910 ft^3/s (table 6) and generally were rated as "good," or within 5 percent of the true discharge. Except for the measurement on July 9, 1998, all measured discharges were within 5 percent of the discharges indicated by the stage-to-discharge relation or had a positive departure from the relation; that is, a greater value than expected from the stage-to-discharge relation. All measurements made on January 8-9, 1998 (the period-of-record peak flow) showed positive departures that are attributed to the uncertainty of the stage-to-discharge relation, which had been extrapolated beyond the previous highest measurement of 1,400 ft^3/s. The measurement on July 9, 1998 indicated 0.68 ft of backwater at the streamflow-monitoring station, but this was a result of debris caught on the branches of a fallen tree that was overhanging the channel about 200 ft downstream. The four negative departures from the stage-to-discharge relation (table 6), which indicated less flow than expected and might have indicated backwater conditions, were within the 5-percent accuracy range of the measurements.

The FCS caused from 0.9 to 1.5 ft of backwater in the wetland immediately upstream of the FCS, depending on discharge, the gate and stop-log configuration of the FCS, and the amount of debris that had caught on the FCS. Extremely high flows were able to pass the FCS on the low, western bank at the Narrows; this decreased the magnitude of backwater that might have resulted from these flows. The results of these studies indicated that (1) the water levels at Blossom Road were controlled by the bridge-opening constrictions of Browncroft Boulevard over Irondequoit Creek and the Millrace, about 5,000 ft upstream from the FCS, and (2) backwater from the FCS did not extend as far upstream as Blossom Road (about 13,000 ft upstream from the FCS).

Plants

Three botanical characteristics of the wetland upstream and downstream from the FCS were evaluated for possible effects of the FCS; these were (1) species richness (number of species) and community composition, (2) cattail productivity, and (3) chemical composition of cattail tissue. Five surveys were conducted, mainly during mid-August of the summers of 1991, 1996, and 1999-2001. The 2000 survey did not include chemical analyses of cattail tissue. Water levels in the wetland were high during the 1991 and 2000 surveys; low during the 1999 survey; and intermediate during the 1996 and 2001 surveys. Observed changes in species richness and cattail-growth characteristics appeared to be related to depth of inundation.

Species Richness and Community Composition

Typha glauca was the dominant species within the wetland, although 59 other plant species in 10 distinct plant communities were identified in 1991 (Bernard and others, 1998). Species richness within the cattail-dominated areas was low; generally the only other major species found among the cattails were floating aquatic plants—primarily duckweeds (*Spirodela polyrhiza*, *Lemna minor*, *Wolffia sp.*). Species richness was greatest in areas of wetness transition, such as along the wetland perimeter, where 34 species were identified in 1991 (Bernard and Seischab, 1991), and on the elevated banks or levees adjacent to the main channel of Irondequoit Creek, where 28 species were found in 1991 (Bernard and Seischab, 1991), and 22 species were found in selected 1 x 2-m plots during 2001 (Noll and others, 2002). These areas differed only slightly in elevation but represented a large wetness gradient over a relatively short distance that created a variety of habitats within a small area. Many of the levee plots were dominated by *Typha glauca*, where an average of 8.2 plant

Table 6. Factors addressed in assessment of hydrologic effects of flow-control structure, Ellison Park wetland, Monroe County, N.Y., 1997-2001

Date	Peak flow (cubic feet per second)	Water-surface profile measured	Backwater at Blossom Road indicated	Discharge measured	Discharge (cubic feet per second)	Departure from stage-discharge relation (percent)
September 29, 1997	962	yes	no	yes	311	-0.2
January 8, 1998	3,300	yes	no	yes	2,900	+7.4
					2,910	+5.4
January 9, 1998	-	no	-	yes	2,600	+8.3
July 9, 1998	1,830	yes	no	yes	814	a
July 10, 1998		no	-	yes	421	-3.4
January 24, 1999	1,900	no	-	yes	692	-2.9
February 25, 2000	782	no	-	yes	746	-0.6
May 13, 2000	2,420	yes	no	no	-	-
March 23, 2001	1,160	yes	no	no	-	-

[a] 0.68 feet of backwater from debris caught on tree overhanging channel about 200 feet downstream from gaging station.

species per plot were identified; in contrast, the cattail-dominated areas along the permanent transects contained only from 1.67 to 2.85 species per plot (table 7; Noll and others, 2002). Species richness was high in the cove area of the wetland, where 23 species were identified in 1999 (Hunter and others, 2000), and an average of 3.78 to 7.88 species per plot were found (table 7). Deep-water areas throughout the wetland were found to contain 19 species of submerged or floating aquatic plants in 1991 (Bernard and Seischab, 1991), whereas selected 1 x 2-m channel plots adjacent to the levee plots contained only 8 species and an average of 2.1 species per plot in 2001 (Noll and others, 2002). Species richness also was low in *Phragmites australis* communities, where only 3 other plant species were found (Bernard and Seischab, 1991).

Species richness and distribution in the wetland were inversely related to depth of inundation of the cattail mat. During wet years, such as 1991 and 2000 (fig. 4), when average water depths at the permanent transects exceeded 0.25 ft, 6 to 7 plant species were identified along the northern transect, and 10 to 14 species were identified along the southern transect (table 7). In contrast, during a dry year, such as 1999, when average water levels were more than 0.65 ft below the top of the cattail mat, 11 and 20 species were found along the northern and southern transects, respectively. The increase in species numbers during 1999 might have resulted from the germination of seeds from annual or short-lived perennial plants—seeds that were dormant in the cattail mat awaiting optimum (drier than normal) conditions for germination and growth (Noll and

Table 7. Measurements obtained in plant survey along three permanent transects in Ellison Park wetland, Monroe County, N.Y., 1991-2001

[FCS, flow-control structure. >, greater than; --, not measured. Transect locations are shown in fig. 7.]

Data category and sampling area	Year				
	1991	1996	1999	2000	2001
Survey Dates					
Transect downstream from FCS	July 5	August 21	August 16	August 16	August 14
Transect upstream from FCS	June 5	August 19	August 17	August 17	August 15
Cove site upstream from FCS	--	August 23	August 18	--	August 16
Number of plots					
Transect downstream from FCS	26	20	30	30	30
Transect upstream from FCS	31	31	29	29	29
Cove site upstream from FCS	0	9	8	0	9
Approximate water-surface elevation at mouth of wetland (feet, North American Vertical Datum, 1988)					
Transect downstream from FCS	247.22	246.96	246.15	247.45	246.51
Transect upstream from FCS	247.86	246.94	246.11	247.51	246.45
Cove site upstream from FCS	--	246.90	246.22	--	246.39
Average water depth at transect (feet; negative number indicates depth to ground water)					
Transect downstream from FCS	.33	.0 [a]	- .66 [a]	.82	.0
Transect upstream from FCS	.36	- .66 [a]	- 1.31 [a]	.26	> - .98
Cove site upstream from FCS	--	--	--	--	1
Number of species					
Transect downstream from FCS	7	4	11	6	7
Transect upstream from FCS	10	9	20	14	16
Cove site upstream from FCS	--	12	23	--	21
Average number of species per plot					
Transect downstream from FCS	2.85	1.75	2.47	1.67	2.03
Transect upstream from FCS	2.45	1.81	2.48	2.07	2.31
Cove site upstream from FCS	--	3.78	7.88	--	6.67
Average number of cattail shoots per 3.3-square-foot (1-square-meter) plot					
Transect downstream from FCS	30.6 ± 1.4	38.3 ± 2.7	37.2 ± 1.7	42.8 ± 1.7	35.3 ± 1.7
Transect upstream from FCS	27.4 ± 2.2	34.8 ± 2.2	33.2 ± 1.8	34.4 ± 2.1	30.7 ± 1.7
Cove site upstream from FCS	--	31 ± 4	35 ± 4	--	25 ± 3
Average height of cattail shoots (feet)					
Transect downstream from FCS	10.0 ± 0.06	9.7 ± 0.23	8.7 ± 0.20	9.4 ± 0.20	8.2 ± 0.30
Transect upstream from FCS	7.1 ± 0.23	9.5 ± 0.16	8.5 ± 0.20	9.6 ± 0.36	8.2 ± 0.23
Cove site upstream from FCS	--	9.1 ± 0.43	9.2 ± 0.23	--	8.9 ± 0.26

[a] Water depth estimated from measured water-surface elevation at mouth of wetland.

others, 2002). These plants were able to complete their life cycle during a single dry growing season. In every survey, fewer species were found along the northern transect than along the southern transect, where the slightly higher land-surface elevation allowed drier conditions than would concurrently prevail in the northern wetland area. Conditions along the cove transect were drier than along either of the other two transects and, thus, showed the greatest species richness during the three years in which all three transects were surveyed (1996, 1999, and 2001; table 7).

Cattail Productivity

Average height, density, and biomass of *Typha glauca* were used as measures of cattail productivity in the wetland (tables 7, 8). Cattail height generally averaged 8 to 9 ft by the mid-August sampling season but could exceed 10.5 ft locally. Cattail density ranged from 25 to 43 shoots per 1-m^2 (3.3 x 3.3-ft^2) plot and averaged 34 shoots/m^2 among all plots in all five survey years. The average weight of an individual cattail shoot (above-ground material) ranged from 25.7 to 90.5 g; the average weight of below-ground material ranged from 33.7 to 108 g per shoot (table 8). The measured weights were greatest during the 1991 and 1996 surveys; those measured after the 1997 FCS installation were less than in 1991 and 1996 along the upstream and downstream transects but were comparable along the cove transect.

Average *Typha glauca* biomass values were computed from the measured values obtained from the samples collected along a given transect during a given year (table 8). These values were compared with those reported in the literature, which reflect seasonal differences within a site, as well as a wide range of climatic and ecological differences among sites. Above-ground biomass of *T. glauca* reported in other studies ranges from 758 to 2,320 g/m^2 (table 9; Van der Valk and Davis, 1978; Bernard and Fitz, 1979; Andrews and Pratt, 1978); shoot biomass of other *Typha* species range from 378 to 2,338 g/m^2 (Whigham and others, 1978; Mason and Bryant, 1975; McNaughton, 1966; Van der Valk and Davis, 1978; Boyd and Hess, 1970; Boyd, 1970, 1971; Bray, 1960; Bray and

Table 8. Cattail biomass in above- and below-ground tissues along the three permanent transects in Ellison Park wetland, Monroe County, N.Y., 1991-2001

[do, ditto; --, not measured; g, grams; kg/m^2, kilograms per square meter. Locations of transects are shown in fig.7.]

Data category and sampling area	Year 1991	1996	1999	2000	2001
Survey dates					
Transect downstream from FCS	August 15 - September 10	July 25 - August 25	August 16	August 16	August 14
Transect upstream from FCS	do	do	August 17	August 17	August 15
Cove site upstream from FCS	--	do	August 18	--	August 16
Number of plots sampled					
Transect downstream from FCS	26	5	10	10	9
Transect upstream from FCS	31	5	9	9	9
Cove site upstream from FCS	0	5	3	0	3
Average above-ground biomass per unit area (kg/m^2)					
Transect downstream from FCS	2.4	3.5	1.3 ± 0.17	1.5 ± 0.20	1.0 ± 0.14
Transect upstream from FCS	2.1	2.5	1.1 ± 0.14	1.3 ± 0.22	1.1 ± 0.14
Cove site upstream from FCS	--	1.0	0.9 ± 0.23	--	1.0 ± 0.28
Average below-ground biomass per unit area (kg/m^2)					
Transect downstream from FCS	3.3	3.6	2.8 ± 0.79	1.6 ± 0.19	1.3 ± 0.19
Transect upstream from FCS	2.7	2.5	1.9 ± 0.31	1.7 ± 0.23	1.5 ± 0.19
Cove site upstream from FCS	--	1.0	1.7 ± 0.56	--	1.4 ± 0.31
Average total biomass per unit area (kg/m^2)					
Transect downstream from FCS	5.7	7.1	4.1 ± 0.81	3.1± 0.78	2.3 ± 0.66
Transect upstream from FCS	4.8	5.0	3.0 ± 0.39	3.0 ± 0.92	2.6 ± 0.65
Cove site upstream from FCS	--	2.0	2.6 ± 0.52	--	2.4 ± 1.2
Average above-ground biomass per shoot (g)					
Transect downstream from FCS	77.4	90.5	34.9	35.0	28.3
Transect upstream from FCS	79.3	72.0	33.0	37.8	35.8
Cove site upstream from FCS	--	32.3	25.7	--	40.0
Average below-ground biomass per shoot (g)					
Transect downstream from FCS	108	94.5	75.3	37.4	36.8
Transect upstream from FCS	102	71.4	57.2	49.4	48.8
Cove site upstream from FCS	--	33.7	48.6	--	56.0

Table 9. Above- and below-ground *Typha* spp. biomass reported in the literature, and average values for each of three permanent transects sampled during four surveys in Ellison Park wetland, Monroe County, N.Y., 1991-2001

Biomass component	Value (grams per square meter)	Location	Reference
Above-ground biomass			
Typha glauca	758 to 2,106	Iowa	Van der Valk and Davis, 1978
	1,361	New York	Bernard and Fitz, 1979
	2,320	Minnesota	Andrews and Pratt, 1978
Other *Typha* species	378 to 1,336	central USA	McNaughton, 1966
	626 to 2,338	Mid-Atlantic coast, USA	Whigham and others, 1978
	428 to 2,252	southeastern USA	Boyd and Hess, 1970
	530 to 1,132	South Carolina	Boyd, 1971
	684	South Carolina	Boyd, 1970
	1,118	England	Mason and Bryant, 1975
	1,070	England	Bray and others, 1959
	1,731	New England	Bray and others, 1959
	1,400	Minnesota	Bray, 1960
Ellison Park *Typha glauca*	900 to 3,500	New York	This study
Below-ground biomass			
Typha glauca	1,167 to 1,450	Iowa	Van der Valk and Davis, 1978
	2,960	Minnesota	Bray, 1960
	2,400 to 3,100	Minnesota	Andrews and Pratt, 1978
Other *Typha* species	556 to 2,646	central USA	McNaughton, 1966
	1,800 to 5,053	Mid-Atlantic coast, USA	Whigham and others, 1978
Ellison Park *Typha glauca*	1000 to 3,600	New York	This study
Total biomass			
Typha spp.	1,289 to 3,405	Oregon	McNaughton, 1966
	972 to 3,982	central USA	McNaughton, 1966
	4,230 to 4,720	Minnesota	Andrews and Pratt, 1978; Bray, 1960
Ellison Park *Typha glauca*	2,000 to 7,100	New York	This study

others, 1959). Average above-ground biomass in the Ellison Park wetland ranged from 900 to 3,500 g/m^2. Below-ground biomass of *T. glauca* reported in other studies ranges from 1,167 to 3,100 g/m^2 (Van der Valk and Davis, 1978; Bray, 1960; Andrews and Pratt, 1978); for other *Typha* species, below-ground biomass ranges from 556 to 5,053 g/m^2 (Whigham and others, 1978; McNaughton, 1966). Average below-ground biomass in the Ellison Park wetland ranged from 1,000 to 3,600 g/m^2. Total biomass, which is infrequently reported in other studies, ranges from 972 to 4,720 g/m^2 (McNaughton, 1966; Andrews and Pratt, 1978; Bray, 1960). The average total biomass in the Ellison Park wetland ranged from 2,000 to 7,100 g/m^2 (table 9). On the basis of these comparisons, the Ellison Park wetland would be considered a highly productive ecosystem.

Above-ground biomass generally represented from 42 to 50 percent of the total biomass, except during the dry 1999 season, when it represented only 32 to 37 percent. The biomass values were highest during 1991 and 1996 and either exceeded or were close to the high end of the range of values reported in the literature (table 9); values measured from 1999 through 2001 generally were within the range of reported values. Biomass was greatest along the northern transect during 1996 and smallest in the cove area during that year. Biomass was greater during the precontrol period than during the postcontrol period; this decline was observed upstream and downstream from the FCS, but not in the cove area (fig. 10). Within-season variations in cattail biomass have been documented in many of the studies referenced above. Annual variation in wetland-plant biomass also has been reported; for example, 47- and 81-percent changes in the total biomasses of two *Typha* spp. communities in Oregon were found over a 1-yr period (McNaughton, 1966), and a 37-percent change in above-ground biomass of *Phragmites australis* was found in Norfolk, England, over a 3-yr period (Boar, 1996). Average total biomass in the Ellison Park wetland varied among survey years—by 68 percent along the northern transect, 48 percent along the southern transect, and 23 percent along the cove transect. These local fluctuations could result from the differences in sampling dates, water depth, or other environmental factors.

Growth rates of emergent macrophytes are related, in large part, to water depth (Coops and others, 1996). Cattail productivity in the Ellison Park wetland, as measured by cattail height, density, and biomass, also could be explained in terms of general soil wetness (Bernard and Seischab, 1991; 1997). Results of the precontrol (1991 and 1996) surveys indicate that cattail productivity was low in dry areas along streambanks and in deep-water areas where water depth exceeded about 1.3 ft (40 cm), and high in areas where the water level was about 1.0 ft (30 cm) above the top of the cattail mat. Data from the three consecutive surveys from 1999 through 2001 indicate an alternative interpretation, however. Cattail productivity during 2001, when water levels were intermediate between the dry conditions of 1999 and the wet conditions of 2000, was lower than in either of the previous 2 years; no direct correlation between water level and productivity within a given year was discernible. The degree of wetness during a given year can directly affect the growth of rhizomes during that year, however, and might indirectly affect productivity during the subsequent year (Noll and others, 2002). Thus, the dry conditions of 1999 might have been conducive to rhizome growth and the late-summer sprouting of numerous shoots, which grew during the 2000 growing season, when wet conditions might have retarded rhizome growth, thus reducing the number of shoots that would be available for growth during the 2001 growing season. This interpretation of the multi-year relation between water level and cattail productivity suggests that wetland studies of 1-yr, and possibly 2-yr, duration might be too short, and lack the necessary data for accurate identification and explanation of this relation.

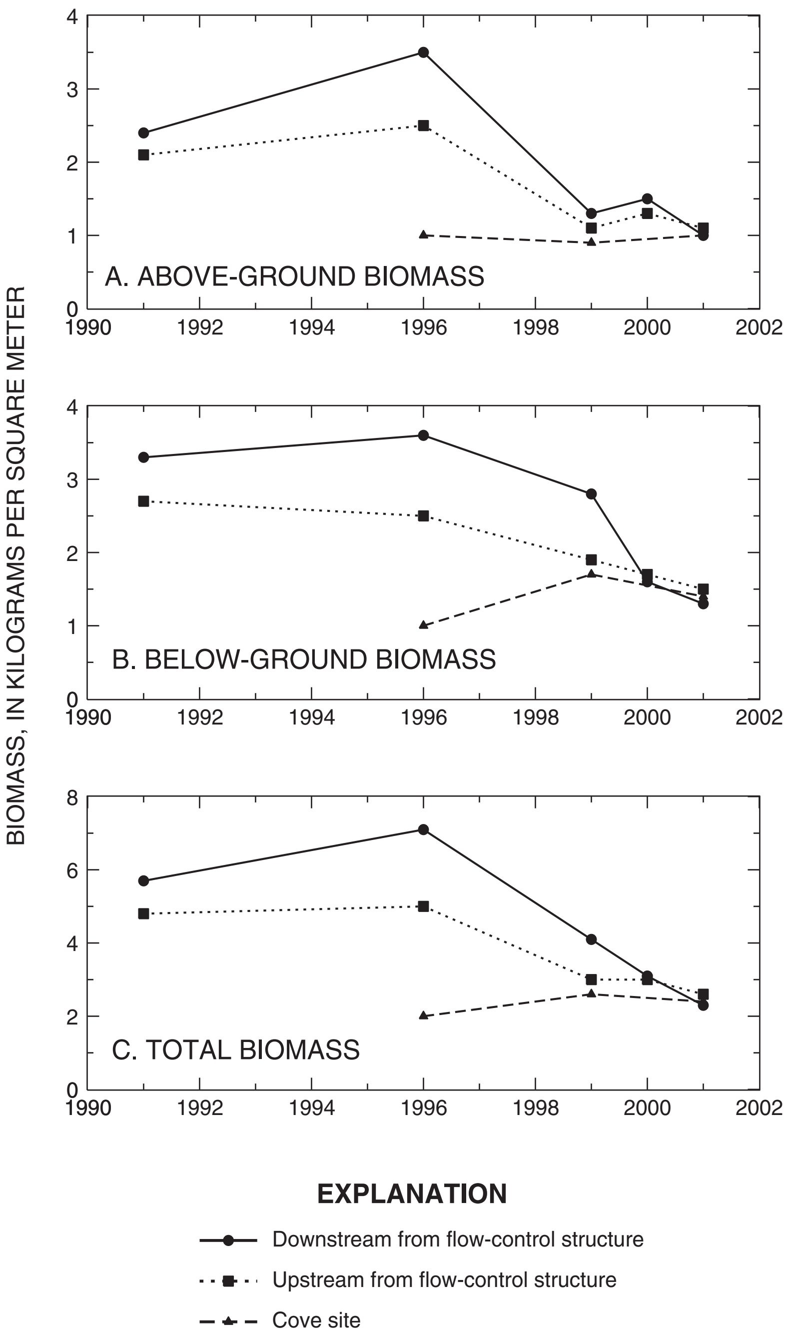


Figure 10. Cattail (*Typha glauca*) biomass measured during five surveys along three permanent transects in Ellison Park wetland, Monroe County, N.Y., 1991-2001

Chemical Composition of Cattail Tissues

Results of the chemical analyses of cattail tissue are presented in table 10; the values represent mean concentrations of major nutrients and trace elements in above- and below-ground tissues collected along the three permanent transects in the wetland during 1991, 1996, 1999, and 2001. Most constituents, except nitrogen, were found at higher concentrations in below-ground tissues than in above-ground tissues; nitrogen concentrations were 2 to 3 times greater in the above-ground tissues. Concentrations of barium, manganese, phosphorus, and potassium were inconsistent from year to year. Aluminum concentrations in below-ground tissue decreased through the study period.

Several constituents showed large differences from year to year, especially iron. The high iron concentrations in below-ground tissues in 1991 and 1996 probably reflected the increased availability of iron, which was released from the sediments under inundated (reduced) conditions in the rhizosphere, and the bioconcentration of iron when close to the thin, oxidized zone surrounding individual cattail rhizomes. The sharp (4-orders-of-magnitude) decrease in iron concentrations from the 1996 to 1999 samples probably reflects the low-water oxidized conditions that would have prevailed in the wetland in 1999; iron would have remained in insoluble forms in the sediment and would not have been available for bioconcentration on the cattail rhizomes. Intermediate water levels in 2001 apparently created less extreme oxidation-reduction conditions and resulted in intermediate iron concentrations in below-ground tissues. Manganese

Table 10. Mean concentrations of selected elements in above- and below-ground cattail tissues collected along three permanent transects in Ellison Park wetland, Monroe County, N.Y., 1991-2001

[Values are in milligrams per kilogram. FCS, flow-control structure; --, not measured; <, less than; bdl, below analytical detection limit. Dates of sample collection are the same as those for biomass samples listed in table 8. No tissue samples were collected from the Cove site in 1991. Locations of transects are shown in fig. 7.]

Element	Downstream from FCS				Upstream from FCS				Cove site		
	1991	1996	1999	2001	1991	1996	1999	2001	1996	1999	2001
Number of samples	26	5	10	9	31	5	9	9	5	3	3
A. Above-ground Cattail Tissues											
Aluminum	15.9	5.0	126	56.0	19.6	7.1	86.7	34.9	7.7	86.7	16.2
Barium	--	--	9.30	14.2	--	--	7.28	8.63	--	16.4	14.9
Boron	7.3	6.1	--	63.5	15.4	7.9	--	247	5.4	--	bdl
Cadmium	bdl	bdl	bdl	bdl	bdl	0.16	bdl	bdl	bdl	bdl	bdl
Calcium	10,000	11,000	12,300	14,700	10,000	6,000	13,700	16,600	10,000	12,800	10,200
Chromium	1.2	0.06	0.97	3.76 [a]	1.5	0.18	0.76	bdl	0.06	0.73	2.72
Cobalt	< 0.1	0.06	0.33	5.35	0.2	0.08	0.42	4.12 [a]	0.06	bdl	6.11
Copper	4.5	4.1	7.33	9.58	5.3	3.8	10.7	8.00 [a]	3.3	5.63	20.4
Iron	58.8	56.9	50.0	35.0	72.3	56.4	46.7	31.6	62.8	40.0	12.6
Lead	0.5	--	0.7 [a]	6.36	0.7	--	0.7 [a]	bdl	--	0.6 [a]	6.46
Magnesium	2,000	2,000	2,180	2,510	1,800	1,600	1,800	1,740	2,000	2,770	1,720
Manganese	478	284	289	247	607	188	244	360	368	378	245
Molybdenum	< 0.1	0.3	0.40	7.01	0.6	0.8	0.66	4.94 [a]	--	0.26	23.7
Nickel	1.6	0.8	0.83	4.66 [a]	1.6	1.3	0.69	bdl	1.2	0.58	11.6
Nitrogen	22,500	22,000	17,400	11,900	26,000	24,000	19,300	12,400	23,000	13,400	11,200
Phosphorus	2,000	2,700	2,890	9,670	2,400	2,900	2,830	8,120	2,500	2,300	7,680
Potassium	11,000	11,000	12,600	1,420	14,000	23,000	6,580	1,050	13,000	3,090	824
Sodium	4,000	--	5,180	15,500	3,490	--	4,760	11,300	--	3,520	7,860
Tin	--	--	248	11.8	--	--	224	14.4	--	240	9.71
Zinc	17.5	13.2	15.8	18.9	15.0	13.9	13.0	16.9	10.4	9.13	12.1
B. Below-ground Cattail Tissues											
Aluminum	1,720	1,200	351	94.8	2,030	1,200	351	138	1,320	290	38.7
Barium	--	--	8.28	9.10	--	--	13.2	9.83	--	11.7	14.3
Boron	53.3	43.2	--	--	212	49.8	--	244	43.5	--	91.6
Cadmium	0.2	bdl	0.61	bdl	bdl	bdl	0.54	0.78 [a]	0.1	bdl	bdl
Calcium	13,000	15,000	8,470	10,700	14,000	13,000	8,790	9,370	12,000	7,960	8,080
Chromium	6.9	3.6	1.12	3.86 [a]	5.2	6.8	1.12	bdl	4.2	0.89	bdl
Cobalt	0.2	0.5	0.52	6.59	1.0	0.7	0.33	5.78 [a]	--	bdl	6.75
Copper	15.5	12.7	13.5	16.4	15.7	15.2	8.70	7.39	11.6	18.2	13.7
Iron	20,300	13,300	1.39	207	15,700	12,400	3.35	1,050	13,900	0.13	65.2
Lead	14.7	--	1.82	9.16 [a]	23.1	--	1.2 [a]	16.1 [a]	--	1.87	bdl
Magnesium	5,000	5,000	3,320	4,510	5,000	6,000	6,430	3,870	7,000	4,770	3,440
Manganese	407	420	86.5	100	508	397	80.9	159	582	104	151
Molybdenum	0.2	0.5	--	15.9	1.0	0.5	--	5.48 [a]	0.2	--	20.8
Nickel	10.9	10.9	0.88	5.93 [a]	8.4	12.6	0.80	bdl	13.1	0.78	13.6
Nitrogen	7,600	7,000	7,500	4,700	9,000	9,000	6,000	5,500	8,000	4,400	6,100
Phosphorus	2,000	2,000	3,160	8,840	2,200	2,000	3,240	10,100	3,000	2,440	8,470
Potassium	5,000	6,000	8,360	1,100	2,000	7,000	7,120	926	9,000	5,880	993
Sodium	10,700	--	10,700	1,680	9,330	--	11,200	13,600	--	7,480	9,170
Tin	--	--	281	3.23	--	--	311	8.84	--	269	4.10
Zinc	46.1	49.7	22.2	93.8	73.6	47.3	16.8	50.9	42.5	12.6	42.0

[a] Estimated value; three or more concentrations were below the analytical detection limit.

concentrations, although lower than iron concentrations, followed a similar pattern through the surveyed years.

Above-ground tissues assimilated an average of 1.12 to 2.6 percent of their dry weight in nitrogen; below-ground tissues assimilated an average of 0.44 to 0.9 percent. Above- and below-ground tissues assimilated similar quantities of phosphorus—an average of 0.2 to 1.0 percent of their dry weight. Generally, upward or downward trends in the concentration data of given constituents from the southern transect were also found in the data from the northern transect. Additional sampling would be needed to detect or confirm any real change in the chemical composition of cattail tissues since the installation of the FCS, however.

Fish

Four characteristics of the fish community upstream and downstream from the FCS were evaluated to discern differences between the two areas; these were (1) species richness and distribution within the wetland; (2) use of the wetland by nonresident species and their distribution; (3) growth rates of selected species; and (4) trophic status of the fish community. Additionally, contaminant-related stress indicators in brown bullhead were identified. Neither electrofishing in 1996 nor video monitoring of fish passage through the FCS in 2001 yielded any results that contributed to the evaluation of the effects of the FCS on the fish community.

Species Richness and Distribution

The Ellison Park wetland is used by a diverse fish community; 44 species were found during the 1991-2001 study period (table 11). Species richness did not change greatly during the study period (Crabtree and Ringler, 2002); each of four surveys identified from 25 to 29 species. The dominant species usually included bluegill and pumpkinseed but also could have included several other species, including alewife, black crappie, brook stickleback, gizzard shad, brown bullhead, and yellow perch. Dominance varied from year to year, or, in some years, from season to season and by wetland area. The dominant species often greatly outnumbered the combined number of all other individuals caught and resulted in an uneven fish community, especially during four periods—May 1992, when 1,066 of 1,833 individuals (58 percent) were alewife; October 1999, when 5,119 of 6,922 individuals (74 percent) were pumpkinseed; July 2001, when 4,070 of 6,446 individuals (63 percent) were yellow perch; and July 1999, when 23,100 of 24,600 individuals (94 percent) were gizzard shad (table 12). Fourteen common species were found during at least one season of each survey year—black crappie, bluegill, bowfin, brown bullhead, common carp, gizzard shad, golden shiner, johnny darter, largemouth bass, northern pike, pumpkinseed, spottail shiner, white perch, and white sucker. Eight rare species were identified by the capture of only one individual of a species during the entire study period; these were American eel (1996), central stoneroller (1991), coho salmon (1996), fathead minnow (1991), freshwater drum (2001), green sunfish (1992), mimic shiner (1992), and rosyface shiner (1992). Brook stickleback and logperch were found during only 1 year's survey—520 brook stickleback were captured in May 1996, and 5 logperch were captured during 2001.

The wetland was used for spawning and nursery habitat, as evidenced by the presence or capture of young-of-the-year (individuals less than 1 year old) and juvenile individuals, adult fishes in breeding colors or with nuptial tubercles, and adult fishes with easily released milt or eggs (Crabtree and Ringler, 2002). In 1991-92, at least 16 species that were confirmed to use the wetland for spawning and(or) rearing habitat, and 6 species that probably did, were identified (Miller and Ringler, 1992). In 2001, 15 species were found as either young-of-the-year, juveniles, or spawning adults. Of these, 10 species were found upstream and downstream from the FCS. The presence of migratory salmonids (brown trout and rainbow trout) indicated that the wetland provides a pathway between Irondequoit Bay and Lake Ontario and spawning areas in Irondequoit Creek upstream of the wetland.

Comparison of the fish-catch size and species richness upstream from the FCS with the same characteristics downstream was possible for 11 of the 12 survey seasons (3 seasons per survey year); extremely low water levels in the upstream wetland area during September 2001 prevented fall sampling that year. Generally, larger catches per unit effort were made in the area downstream from the FCS than in the area upstream (table 12), especially during those seasons when the catch size was affected by a large influx of fish from Irondequoit Bay (May 1992, July 1999, and July 2001). Upstream catches exceeded downstream catches during July and September 1991, September 1996, and June 1999. Fewer species were found upstream from the FCS than downstream during 5 of the 11 surveyed seasons; more species, or an equal number of species, were found upstream than downstream during the other six seasons (table 12). The FCS did not appear to affect the relations between upstream and downstream fish-catch size or species richness.

Resident- and Nonresident-Fish

Of the 44 species found in the wetland, 25 were considered resident species—those that are likely to spend all of their lives within a marsh setting (table 11) according to the classification of Jude and Pappas (1992). The remaining 19 species were considered nonresident species—those that migrate into a marsh environment for only part of their life cycle. The resident species outnumbered the nonresident species in all survey years. The number of resident species found in the wetland ranged from 15 to 17, whereas the number of nonresident species ranged from 14 in 1991 to 10 in 1999 (table 11). The number of nonresident species found in the area upstream from the FCS during the precontrol period ranged from 0 to 4 less than the number found in the area downstream from the FCS

Table 11. Fish species found in Ellison Park wetland, Monroe County, N.Y., 1991-2001

Species		Wetland resident species	July and September 1991	May 1992	1996	1999	2001
Common name	Scientific name						
Longnose gar	*Lepisosteus osseus*	yes		X	X	X	
Bowfin	*Amia calva*	yes	X	X	X	X	X
American eel	*Anguilla rostrata*	no			X		
Alewife	*Alosa pseudoharengus*	no	X	X	X		X
Gizzard shad	*Dorosoma cepedianum*	yes	X	X	X	X	X
Goldfish	*Carassius auratus*	yes		X	X	X	
Common carp	*Cyprinus carpio*	no	X	X	X	X	X
Central stoneroller	*Campostoma anomalum*	yes	X				
Spotfin shiner	*Cyprinella spiloptera*	yes	X	X			X
Common shiner	*Luxilus cornutus*	yes			X	X	
Emerald shiner	*Notropis atherinoides*	no	X			X	X
Spottail shiner	*Notropis hudsonius*	no	X	X	X	X	X
Rosyface shiner	*Notropis rubellus*	yes		X			
Mimic shiner	*Notropis volucellus*	no		X			
Bluntnose minnow	*Pimephales notatus*	yes	X	X		X	X
Fathead minnow	*Pimephales promelas*	yes	X				
Golden shiner	*Notemigonus crysoleucas*	yes	X	X	X	X	X
Rudd	*Scardinius erythrophthalmus*	yes	X	X	X		X
Quillback	*Carpiodes cyprinus*	yes		X			X
White sucker	*Catostomus commersoni*	no	X	X	X	X	X
Shorthead redhorse	*Moxostoma macrolepidotum*	no	X				X
Brown bullhead	*Ameiurus nebulosus*	yes	X	X	X	X	X
Channel catfish	*Ictalurus punctatus*	no				X	X
Northern pike	*Esox lucius*	no	X	X	X	X	X
Central mudminnow	*Umbra limi*	yes	X		X	X	X
Coho salmon	*Oncorhynchus kisutch*	no			X		
Rainbow trout (steelhead)	*Oncorhynchus mykiss*	no		X	X		X
Brown trout	*Salmo trutta*	no	X	X			
Trout perch	*Percopsis omiscomaycus*	no	X	X			
Brook silverside	*Labidesthes sicculus*	no	X	X			
Banded killifish	*Fundulus diaphanus*	yes		X		X	X
Brook stickleback	*Culaea inconstans*	yes			X		
White perch	*Morone americana*	yes	X	X	X	X	X
Rock bass	*Ambloplites rupestris*	yes	X		X	X	X
Green sunfish	*Lepomis cyanellus*	yes		X			
Pumpkinseed	*Lepomis gibbosus*	yes	X	X	X	X	X
Bluegill	*Lepomis macrochirus*	yes	X	X	X	X	X
Smallmouth bass	*Micropterus dolomieu*	no	X			X	
Largemouth bass	*Micropterus salmoides*	yes	X	X	X	X	X
Black crappie	*Pomoxis nigromaculatus*	no	X	X	X	X	X
Johnny darter	*Etheostoma nigrum*	no	X	X	X	X	X
Yellow perch	*Perca flavescens*	no	X		X	X	X
Logperch	*Percina caprodes*	yes					X
Freshwater drum	*Aplodinotus grunniens*	yes					X
Total number of species			**29**	**29**	**26**	**25**	**29**
Number of resident species / number of nonresident species			**15 / 14**	**17 / 12**	**15 / 11**	**15 / 10**	**17 / 12**

Table 12. Results of fish surveys in Ellison Park wetland, Monroe County, N.Y., 1991-2001

[FCS, flow-control structure.]

Fish-community characteristic	1991-92			1996		
	May 1992[a]	July 1991	Sept. 1991	May	July	September
A. PRECONTROL PERIOD						
Number of individuals caught						
Area downstream from FCS	1,439	344	217	954	398	94
Area upstream from FCS	394	417	253	115	138	120
Number of species identified: resident / nonresident						
Area downstream from FCS	14 / 9	11 / 9	8 / 10	12 / 6	7 / 7	5 / 6
Area upstream from FCS	14 / 9	11 / 9	7 / 7	6 / 2	6 / 5	6 / 6
Most abundant species						
Area downstream from FCS	Alewife	Bluegill	Pumpkinseed	Brook stickleback	Alewife	Pumpkinseed
Area upstream from FCS	Alewife	Black crappie	Bluegill	Brook stickleback	Pumpkinseed	Bluegill
Other abundant species (more than 20 individuals caught), in order of abundance						
Area downstream from FCS	White perch Gizzard shad Pumpkinseed Brown bullhead Spottail shiner	Brown bullhead Alewife Common carp Pumpkinseed Gizzard shad	Gizzard shad Brook silverside	Pumpkinseed Bluegill Black crappie	Pumpkinseed Bluegill	
Area upstream from FCS	Pumpkinseed Gizzard shad White perch Bluegill Brown bullhead	Gizzard shad Bluegill Largemouth bass	Bluntnose minnow			Pumpkinseed
	1999			**2001**		
	June	July	October	May	July	September
B. POSTCONTROL PERIOD						
Number of individuals caught						
Area downstream from FCS	106	23,692	4,348	347	6,209	3,119
Area upstream from FCS	265	965	2,574	217	237	---[b]
Number of species identified: resident / nonresident						
Area downstream from FCS	7 / 5	10 / 7	11 / 6	9 / 7	13 / 8	13 / 6
Area upstream from FCS	8 / 5	9 / 4	10 / 8	10 / 8	5 / 6	---
Most abundant species						
Area downstream from FCS	Pumpkinseed	Gizzard shad	Pumpkinseed	Brown bullhead	Yellow perch	Pumpkinseed
Area upstream from FCS	Pumpkinseed	Gizzard shad	Pumpkinseed	Pumpkinseed	Pumpkinseed	---
Other abundant species (more than 20 individuals caught), in order of abundance						
Area downstream from FCS		Pumkinseed White sucker Emerald shiner Common carp Yellow perch Largemouth bass Spottail shiner Bluegill Golden shiner	Bluegill Goldfish Largemouth bass Yellow perch Common carp White sucker Golden shiner	Pumpkinseed Emerald shiner	Brown bullhead Pumpkinseed Golden shiner White sucker Bluntnose minnow Common carp Bluegill	Yellow perch Bluegill Golden shiner Gizzard shad
Area upstream from FCS	Emerald shiner Bluegill Gizzard shad Golden shiner	Pumpkinseed Common carp Largemouth bass Bluegill White sucker	Bluegill Gizzard shad Common carp Yellow perch Largemouth bass Emerald shiner	Brown bullhead Bluegill	Golden shiner Yellow perch	

[a] May 1992 data are listed before 1991 data to maintain seasonal uniformity among the surveys.

[b] Extremely low water levels prevented use of trap nets.

(table 12). This relation was found during 3 of the 5 sampling seasons during the postcontrol period, but the number of nonresident species in the upstream area exceeded that in the downstream area during October 1999 and May 2001.

The wetland functioned as two different systems—the southern wetland area (upstream from the FCS) was dominated by resident species, whereas the northern area contained a mixture of residents and nonresidents, with a pulse of nonresidents (either spawners or young-of-the-year) during one season of each surveyed year (Crabtree and Ringler, 2002). The dominant nonresident species was alewife in May 1992 and July 1996; and yellow perch in 1999 and 2001 (the number of yellow perch caught in 2001 was 10 times that in 1999). These differences, as well as the fluctuations in total catch among years and seasons, probably reflected natural or background population dynamics throughout the Irondequoit Bay system, and possibly the greater southern Lake Ontario near-shore system, rather than the effects of the FCS (Crabtree and Ringler, 2002).

Growth Rates of Selected Species

Growth rates of five species—black crappie, bluegill, largemouth bass, northern pike, and pumpkinseed—were calculated for each of the survey years (table 13). The most rapid growth for all species occurred during the first or second year. All of these species showed large year-to-year variations in growth rate, especially northern pike, bluegill, and pumpkinseed. Largemouth bass showed the most consistency in growth rates among the surveyed years. Statistically significant differences in growth rates were detected between certain years for some species (Thompson and others, 2000), but these differences could not be attributed to the FCS (Crabtree and Ringler, 2002). The fluctuations in growth rates probably result

Table 13. Growth rates of selected fish species in Ellison Park wetland, Monroe County, N.Y., 1991-2001

[Values are in grams per year. DS (US), downstream (upstream) from flow-control structure.]

Species	Year collected	Age, in years 0-1	1-2	2-3	3-4	4-5	5-6	6-7	7-8	8-9
Black crappie	1991	1.96	0.81	0.57	0.42	0.39	0.37	0.14	0.17	
	1996	2.52	1.05	.57	.27					
	1999 - DS	1.28	1.43	.60						
	1999 - US	1.46								
	2001	2.43	.86	.71	.33	- .12	- .11	.52	.52	.23
Bluegill	1991	.68	1.56	.76	.44	.33	.29			
	1996	.69	1.32	.72	.57	.39	.30	.17	.14	.10
	1999 - DS	3.83	1.71	.88	.46	.38	.61	.24		
	1999 - US	- .78	.03	.016	.007	.006	.004	.002		
	2001 - DS	1.43	1.36	.95	.34	.15	.07	.72		
	2001 - US	2.77	.65	.50	.24	.17	- .01			
Largemouth bass	1991	1.79	1.04	.69	.45	.35	.43	.19	.30	
	1996	1.45	1.16							
	1999 - DS	1.21	.90							
	1999 - US	1.29	1.17	.62	.44					
	2001	Growth rates not computed								
Northern pike	1991	4.06	2.60	1.22	.87					
	1996	4.33	1.87	1.23	.50					
	1999 - DS	8.61	.043	.043	.027	.008	.007	.013		
	1999 - US	7.10	.008	.004	.003	.002				
	2001	Growth rates not computed								
Pumpkinseed	1991	.98	1.14	.57	.94	.52	.30			
	1996	.88	1.14	.84	.56	.48	.37	.39		
	1999 - DS	.32	1.61	.88	.36	.30	.18	.28	.15	
	1999 - US	.38	1.86	.80	.44	.43	.19	.16		
	2001 - DS	2.77	1.07	.66	.18	.30				
	2001 - US	1.72	.97	.60	.38	.50				

from natural changes in annual temperature and prey or predator densities.

Trophic Status of Fish Community

The trophic status of the Ellison Park wetland community changed seasonally and from year to year. Insectivores dominated the wetland during most seasons and represented 88 percent of the fish found in the northern wetland area during May 1996 (table 14). Invertivores dominated the northern wetland area during May 1992 and July 1996 because of an unusually large influx of nonresident alewife. Omnivores, which usually represented less than 30 percent of the fish community, dominated the northern and southern wetland areas during July 1999, when gizzard shad were found in extremely large numbers. Piscivores, such as walleye and trout, were relatively rare (usually less than 15 percent of the fish community), probably because these are cool-water fish that do not tolerate the warm water temperatures of the wetland for prolonged periods (Thompson and others, 2000), but also because piscivores typically represent a small percentage of a fish assemblage (D.B. Chambers, U.S. Geological Survey, written commun., 2002). This trophic class dominated the northern wetland area during July 2001 with the large influx of yellow perch; these fish were young-of-the-year, however, and less piscivorous than adult yellow perch (D.B. Chambers, U.S. Geological Survey, written commun., 2002). Use of generalized trophic classes such as used in these fish studies could oversimplify, and possibly misrepresent, the Ellison Park wetland community (D.L. Crabtree, State University of N.Y., College of Environmental Science and Forestry, written commun., 2002).

Apparent shifts in the trophic status of the wetland were strongly related to the presence of a dominant fish species or a combination of nondominant species that were found in fairly equal numbers in the wetland. The fish community in the southern wetland area (upstream from the FCS) during May 1992 included many species in fairly equal numbers; the trophic-group percentages clustered between 14 and 34 percent (table 14). Conversely, the fish community in the northern wetland area during the same season was dominated by a single species (alewife), and the resulting trophic status of that wetland area reflected the trophic class of alewife, that is, invertivores. Similarly, brook stickleback greatly affected the trophic status of the wetland during May 1996, as did gizzard shad during July 1999.

Apparent changes in trophic composition of the fish community either from year to year, seasonally, or within a given area of the wetland, were probably related to the effect of a dominant species, the availability of prey, or regional trends in fish-population sizes. The data from the four fish surveys did not identify any shift in the trophic status of the fish community in the southern wetland area that was not seen in the northern wetland area or that could be attributed directly to the FCS, or indirectly to changes in sediment- and nutrient-accumulation rates.

Table 14. Trophic status of fish community in Ellison Park wetland, Monroe County, N.Y., 1991-2001

[Values are percentage of total catch in each trophic group. DS (US), downstream (upstream) from flow-control structure. --, no data.]

Season	Sampling area	Trophic Group			
		Insectivore	Invertivore	Omnivore	Piscivore
July 1991	DS	34	36	17	13
	US	26	36	29	9
September 1991	DS	58	0	23	19
	US	74	2	18	6
May 1992	DS	11	74	7	8
	US	34	27	25	14
May 1996	DS	88	5	5	2
	US	82	3	12	3
July 1996	DS	29	64	4	3
	US	72	5	9	14
September 1996	DS	79	2	13	6
	US	77	2	10	11
June 1999	DS	75	1	12	11
	US	63	6	24	6
July 1999	DS	4	0	94	2
	US	26	1	67	7
October 1999	DS	86	0	7	7
	US	84	0	8	7
May 2001	DS	78	9	6	7
	US	61	20	9	10
July 2001	DS	30	1	3	66
	US	57	5	24	14
September 2001	DS	56	20	4	20
	US	--	--	--	--

Contaminant-Related Stress Indicators in Brown Bullhead

Contaminant-related stress indicators in brown bullheads were noted during the July 1991, May 1992, and 1999 sampling periods. Stress indicators in bottom-feeding fish result from sediment contamination by organic compounds and metals (Hickey and others, 1990; Hirethota and Ringler, 1992). Of the 164 brown bullheads captured during the 1991-92 survey, 128 were examined for melanoma, tumors, lesions, and barbel deformities (shortened, bent, burnt, bumpy, or missing). Of these, about 75 percent showed contaminant-related stress indicators (table 15). All 38 individuals captured during the 1999 survey showed melanotic pigmentation, lesions, oral

Table 15. Indicators of contaminant-related stress found in brown bullheads in Ellison Park wetland, Monroe Country, N.Y., 1991-2001

[Stress indicators not recorded during 1996 or 2001 surveys.]

Indicator	Frequency of occurrence (percentage of fish examined) July 1991	May 1992	1999
Number of fish examined	**23**	**105**	**38**
Bent barbel	21.7	6.7	26.3
Bumpy barbel	0.0	10.5	5.3
Burnt barbel	43.5	41.9	--
Missing barbel	8.7	7.6	7.9
Shortened barbel	4.3	1.0	10.5
Lesion or ulcer	26.0	25.7	31.6
Melanoma	26.0	14.3	57.9
Tumor or raised-skin growth	13.0	7.6	21.0
No symptoms found	8.7	29.5	0.0

ulcers, bent barbels and other indicators of stress (table 15). No note of contaminant-related stress indicators was made on brown bullheads that were captured during the 1996 and 2001 surveys.

The most common indicator noted in 1991 and 1992 was burnt barbel, which was found on 44 to 42 percent of the 128 fish examined, respectively. This indicator was not recorded during the 1999 survey, but other barbel deformities were found on at least 26 percent of the fish captured that year. During 1999, the most prevalent indicator was melanoma (skin pigmentation), which was noted on 22 (58 percent) of the captured fish. The incidence of contaminant-related stress indicators appeared to be related to fish size and location in the wetland; large fish in the northern part of the wetland (close to Irondequoit Bay) showed more signs of stress than small fish in that area and more signs than fish in the southern part of the wetland. Some compounds that could be the cause of these indicators in brown bullhead, such as polycyclic aromatic hydrocarbons, have been found in high concentrations in the wetland sediments. (See “Sediment Quality” section, further on.) The actual source and extent of the contaminant-related stress indicators in brown bullheads are obscured by the proximity of Irondequoit Bay, into and from which brown bullhead can freely migrate; however, the FCS probably had no direct relation to the prevalence of these stress indicators in the wetland (Thompson and others, 2000).

Turtles

The capture of fish often resulted in the capture of turtles as well, mostly in the trapnets (table 16). The only two species found during all sampling seasons were painted and snapping turtles; two spiny softshell turtles were caught in 1999. As many as 208 painted turtles (1999) and 105 snapping turtles (1996) were caught in a year. Generally, more turtles of both species were caught in the southern wetland area (upstream from the FCS) than in the northern area. During 1991, 115 turtles were individually marked with notches filed in their scutes. During May 1992, 12 of the 91 marked painted turtles from the southern wetland area were recaptured; this allowed the painted-turtle population of the southern wetland area to be estimated at 478 ± 227. No turtles were recaptured in the northern wetland area. The spiny softshell turtle (*Trionyx spiniferus*) is listed as a New York State species of concern and was not known to inhabit the Ellison Park wetland (Thompson and others, 2000).

Benthic Macroinvertebrates

No benthic macroinvertebrate survey was conducted after installation of the FCS; therefore, the effect of the FCS on the benthic-macroinvertebrate community could not be assessed. Nonetheless, the macroinvertebrate data are presented to characterize, in combination with other survey results, the precontrol ecological condition of the wetland.

During 1991-92, the benthic macroinvertebrate community of the wetland was characterized through an analysis of the stomach contents of 99 individual fish of 9 species (Miller and Ringler, 1992). Thirteen taxa of macroinvertebrates were identified, generally to the lowest taxonomic level possible. The most abundant taxa was Cladocera (crustaceans—*Daphnia*, *Bosmina*), which represented 87.4 percent of all macroinvertebrates identified. The predominant insect family was Chironomidae (midges), which represented 82.8 percent of all insects, and 9.6 percent of all macroinvertebrates identified by this method. Other insect taxa included Trichoptera (caddisflies), Plecoptera (stoneflies), Zygoptera (damselflies), Veliidae (water striders),

Table 16. Turtle species found in Ellison Park wetland, Monroe County, N.Y., 1991-2001

[US (DS), upstream (downstream) from flow-control structure.]

Species		July 1991	May 1992	1996		1999		2001	
Common name	Scientific name			US	DS	US	DS	US	DS
Painted turtle	*Chrysemys picta*	129	76	187	6	129	79	81	32
Snapping turtle	*Chelydra serpentina*	46	16	90	15	43	52	52	6
Spiny softshell turtle	*Trionyx spiniferus*	0	0	0	0	2	0	0	0

Table 17. Benthic macroinvertebrates found in Ellison Park wetland, Monroe County, N.Y., September-October 1996
[Data from Haynes and McNamara (1998). P., Phylum; C., Class; O., Order; F., Family.]

Taxon	Creek sites (sediment and vegetation) Southern area	Creek sites (sediment and vegetation) Northern area	Wetland sites (sediment and vegetation) Southern area	Wetland sites (sediment and vegetation) Northern area	Open-water sites	Number in taxon	Percentage of total in taxon
P. Platyhelminthes	14	3	7	0	4	28	0.07
P. Nematomorpha	1	2	1	2	0	6	.01
P. Annelida							
C. Oligochaeta	740	2,720	2,228	1,243		6,931	16.06
Other Annelida	0	0	10	2		12	.03
P. Arthropoda							
C. Crustacea							
O. Amphipoda	10,256	5,239	10,839	5,369	160	31,863	73.85
Other Crustacea	118	153	527	143	53	994	2.30
C. Insecta							
O. Diptera							
F. Chironomidae	201	376	296	237	1	1,111	2.58
Other Insecta	366	205	584	124	8	1,287	2.98
C. Arachnida	27	23	3	3	0	56	.13
P. Mollusca							
C. Gastropoda	258	198	133	113	28	730	1.69
Other Mollusca	2	31	88	10	0	131	.30
Subtotal, by area	11,983	8,950	14,716	7,246	254	43,149	
Percentage of total, by area	27.77	20.74	34.11	16.79	0.59		100.00

Hydrachnidia (water mites), and Circulionidae (weevils). Other taxa identified were Copepoda (copepods), Amphipoda (scuds), Oligochaeta (segmented worms), and Astracidae (crayfish). This sampling method provided only general information on the macroinvertebrate community; the results cannot be considered comprehensive. The actual macroinvertebrate composition would be more diverse and also would change seasonally, depending on the availability of particular macroinvertebrates as food sources and on the selectivity of feeding fish (Miller and Ringler, 1992).

During September and October 1996, macroinvertebrate samples were collected from three habitats—channel bottom, vegetation, and pelagic zone—at one creek-channel site and one wetland (backwater channel) site in each of the southern and northern parts of the wetland (Haynes and McNamara, 1998). Only 254 individuals from 12 species were found on the multiplate samplers in the pelagic (open-water) habitats; therefore, the data from this habitat are not included in the following analyses.

Amphipods dominated the macroinvertebrate community at all four locations sampled and represented 74 percent of all macroinvertebrates collected (table 17). The only other abundant macroinvertebrate group found in the wetland was Oligochaeta, which represented 16 percent of all macroinvertebrates collected. Other groups found included nonamphipod Crustacea (2.3 percent), Chironomidae (2.6 percent), nonchironomid Insecta (3.0 percent), and Gastropoda (mollusks, 1.7 percent). Physical characteristics measured at the two creek sites and two wetland sites to assess comparability of the macroinvertebrate data (table 18) indicated that some factors, such as water temperature, specific conductance, dissolved-oxygen concentration, pH, percent canopy cover, and flow velocity, were similar, whereas others, such as water depth, amount of vegetation, and substrate composition, differed. In general, no major differences were noted; thus, comparison of the creek and wetland habitats in the southern part of the wetland with those in the northern part was considered valid.

Community Indicators of Water Quality

The benthic-macroinvertebrate taxa found in the Ellison Park wetland during the 1996 survey were typical of slow-moving, organically rich, oxygen-poor aquatic habitats with low taxonomic diversity and large populations of species that

Table 18. Physical characteristics of sampling sites and benthic-macroinvertebrate measures of ecological health in southern and northern areas of Ellison Park wetland, Monroe County, N.Y., September-October 1996

[Data from Haynes and McNamara (1998). µS/cm at 25 ˚C, microsiemens per centimeter at 25 degrees Celsius.]

Site characteristics and measures of ecological health	Creek sites		Wetland (backwater-channel) sites	
	Southern area	Northern area	Southern area	Northern area
Physical Characteristics of Sites				
Dominant substrate material	Sand	Mud	Mud	Mud
Water depth (feet)	1.80	3.08	1.94	2.00
Channel width (feet)	60	60	26	46
Water temperature (degrees Celsius)	14.9	10.5	16.3	15.7
Specific conductance (µS/cm at 25 ˚C)	600	555	540	600
Dissolved oxygen (milligrams per liter)	6.9	12.6	6.1	5.0
pH	8.1	7.8	7.7	8.0
Percentage of area covered by submerged vegetation	80	60	80	0
Flow velocity (feet per second)	0.95	0.59	0.10	0.20
Percent canopy cover	2	0	0	0
Measures of Ecological Health				
Substrate habitat				
Number of organisms collected	879	2,856	1,732	1,851
Species richness (no. of species or taxa in a sample)				
number of species	18	19	17	20
species with more than 1 individual	12	14	13	15
species with more than 4 individuals	9	10	9	9
Simpson s diversity index [1]	0.566	0.399	0.418	0.629
Percent model affinity (PMA) [2]	46	42	39	44
Dominance-3 model (DOM-3) [3]	92	97	96	91
Hilsenhoff biotic index (HBI) [4]	8.13	9.41	8.59	7.21
Qualitative impact assessment [5]	moderate	moderate to severe	moderate to severe	moderate
Vegetation habitat				
Number of organisms collected	11,104	6,094	12,984	5,395
Species richness				
number of species	50	38	50	43
species with more than 1 individual	36	27	37	25
species with more than 4 individuals	17	15	27	13
Simpson s diversity index [1]	0.414	0.431	0.530	0.451

[1] Simpson s diversity index combines species richness and community balance (evenness) to quantify ecological health of a habitat. 0 = highly diverse, 1.0 = nondiverse (single-species dominance)

[2] PMA, a measure of community-composition similarity to an unaffected community. Values > 55 indicate little adverse effect.

[3] DOM-3, a measure of community balance evenness of distribution of individuals among the species. DOM-3 is the combined percent contribution of the three most numerous species. High values (approaching 100) indicate unbalanced communities.

[4] HBI, a combined measure of species abundance and tolerance. Scale is 1 to 10; low values indicate clean-water conditions.

[5] Qualitative terms, such as severe, moderate, slight, and none, can be used to describe the level of ecological impact that is quantitatively defined by the biotic indices.

can survive under these conditions (Haynes and McNamara, 1998). Oligochaetes (segmented worms, mainly *Limnodrilus hoffmeisteri*) were most abundant in the muddy sediment samples, whereas crustaceans (mainly amphipods, *Gammarus spp.*) dominated submerged-vegetation habitats. Several measures of ecological health, such as species richness, and biotic indices such as Simpson's diversity index, percent model affinity, dominance, and Hilsenhoff's biotic index (Bode and others, 1996), were calculated (table 18). Species richness is the total number of species or taxa found in a community; high values (greater than about 20) are mostly associated with clean-water conditions. Species richness in the channel-bottom (substrate) habitats ranged from 17 to 20, but these values were halved when only those species represented by more than four individuals were counted. Species richness was greater in the submerged-vegetation habitats than the substrate habitats; values ranged from 38 to 50 but decreased to 13 to 27 if only species represented by more than four individuals were counted.

A species-diversity index, such as Simpson's index, combines species richness and community balance (evenness) to quantify the ecological quality of a habitat. Simpson's diversity index, which is the probability that two individuals, chosen at random and independently from the population, will be found to belong to the same group, can range from 0 for a highly diverse and balanced community to 1.0 for a nondiverse community that is dominated by one or only a few species (Washington, 1984). Simpson's diversity index ranged from 0.399 to 0.629 in the substrate habitats and from 0.414 to 0.530 in the vegetation habitats. Percent model affinity (PMA) is a measure of similarity to a nonimpaired model community, as described by the percent abundance of seven major taxonomic groups; values greater than 55 percent indicate similarity and good ecological quality. Wetland substrate PMA values ranged from 39 to 46. Dominance is a measure of community balance, or evenness of the distribution of individuals among the species; high values that approach 100 indicate unbalanced communities that are strongly dominated by one or only a few species. Dominance-3 model (DOM-3) is the combined percent contribution of the three most numerous species; DOM-3 values ranged from 91 to 97 in the substrate habitats. The Hilsenhoff biotic index (HBI) is a combined measure of species abundance and tolerance to pollution; on a scale of 0 to 10, low values are indicative of clean-water conditions. Wetland substrate HBI values ranged from 7.21 to 9.41. These biotic index values reflect the macroinvertebrate-community's response to many environmental factors, including habitat and sediment quality, as well as water quality, and indicate that the ecological health of the Ellison Park wetland is moderately to severely impaired (table 18; Haynes and McNamara, 1998).

Benthic-macroinvertebrate community differences between the southern and northern wetland areas were summarized by Haynes and McNamara (1998) as follows:

1. The sand substrate of the southern creek site contained fewer species and organisms per sample than the mud substrate of the northern creek site, but the southern creek site had better balance among species and, thus, better ecological quality than the northern creek site as indicated by biotic-index values.
2. Conversely, the vegetation-habitat samples from the southern creek site showed greater species richness and higher organism counts than those from the northern creek site, but the evenness component of the diversity index indicated slightly better ecological quality in the northern-creek vegetation habitat.
3. Of the two wetland backwater sites, which both had mud substrates, the southern site had lower species richness and lower numbers of organisms than the northern site, and much lower biotic-index values than the northern site.
4. Of the two backwater vegetation-habitat sites, the southern site was far more densely vegetated than the northern site, and had greater species richness, higher organism counts, and a greater diversity value.
5. Differences between the macroinvertebrate communities in the southern creek and wetland sites and those in the northern sites were probably due to habitat differences, such as substrate composition, water depth, and density of submerged aquatic vegetation.

Species Indicators of Water Quality

Among the taxa of benthic macroinvertebrates that could be used as indicators of current water-quality conditions and of future water-quality changes in the Ellison Park wetland are *Gammarus spp.* (scuds or sideswimmers) and chironomids (Haynes and McNamara, 1998). Pollution-tolerance criteria for macroinvertebrate taxa range from 0 for highly intolerant of pollution to 10 for highly tolerant (Bode and others, 1996). A high ratio of *Gammarus pseudolimnaeus*, which has a tolerance value of 4, to *G. fasciatus*, which has a tolerance value of 6, is indicative of good water quality. Both species are abundant in the Ellison Park wetland; *G. pseudolimnaeus* (27,040 individuals) represented 85 percent of the genus, and *G. fasciatus* (4,823 individuals) represented the other 15 percent. The fairly high ratio of 5.67 for these two species indicates good water quality in the wetland, but an increase in numbers of *G. fasciatus* might indicate deterioration in water quality.

Another indicator of environmental pollution is deformed mouthparts in chironomids (Warwick, 1990). Among chironomid taxa, *Procladius* is considered to be more tolerant of pollution than *Chironomus* (Warwick, 1989). Of 51 *Procladius* individuals collected in the wetland, only one was deformed, and of 392 *Chironomus* individuals, 30 (7.6 percent) had mouthpart deformities. Other studies have concluded that deformity rates of less than 14 percent for chironomids (Warwick, 1990), and less than 12 percent for *Chironomus* (Lenat and Barbour, 1994) are typical in organically enriched sediments characterized by low dissolved-oxygen concentration and high biological oxygen demand, such as in the Ellison Park wetland. A deformity rate greater than 20 percent would indicate exposure to toxic chemical pollution (Lenat and Barbour, 1994). Less than 4 percent of the chironomids collected, and 7.6 percent of *Chironomus,* had mouthpart deformities; therefore, the observed incidence of deformities is attributed to exposure to organic enrichment of the sediment rather than to toxic chemicals (Haynes and McNamara, 1998).

Fish-Stomach-Contents Data

Benthic macroinvertebrates found in the stomach contents of 40 fish—20 bluegill (*Lepomis macrochirus*) and 20 pumpkinseed (*L. gibbosus*)—that were captured by electrofishing during September and October 1996, were compared with those identified in the stomach contents of 7 bluegill and 8 pumpkinseed captured during July or September 1991 or May 1992. The diets of these species differed sharply between the two sampling years—1991-92 stomach contents were dominated by chironomids (midges), and even within that survey period, bluegill preferred fish eggs over any macroinvertebrate during May 1992; whereas 1996 stomach contents were dominated by amphipods (scuds). In addition to the discrepancies in sample sizes, fish-capture methods, and sampling seasons, these differences were probably a result of natural variability in the dominant food species for these fish. The large percentage of amphipods found in the 1996 stomach contents—92 percent of all macroinvertebrates identified —reflected the abundance of amphipods—74 percent of all macroinvertebrates identified—throughout the wetland. Amphipods made up less than 3 percent of the total diet of bluegill and pumpkinseed, and less than 1 percent of the stomach contents of all fish analyzed in 1991-92, however. In contrast, the preferred food source of bluegill and pumpkinseed in 1991-92 (chironomids) represented less than 10 percent of the macroinvertebrates identified in stomach contents of all fish examined during that survey, and did not reflect the wetland-wide dominance of Cladocera, which was the preferred food source of black crappie. Large numbers of fish eggs consumed during May 1992 simply reflected the abundance of that food source during spawning season.

Birds

Three surveys were conducted in the wetland (1991-92, 1996, and 2002) to identify the presence and absence of bird species during the pre- and postcontrol periods. An earlier (1980-85) statewide survey by the New York State Department of Environmental Conservation and the Federation of New York State Bird Clubs to identify bird species and map the extent of their breeding ranges in New York provided an initial baseline listing of birds that were likely to be found in or around the wetland during the early 1980’s (Andrle and Carroll, 1988). The data collected during that study were compiled by survey blocks, each of which covered about 10.5 mi^2 in the Ellison Park area. The wetland, which occupies less than 1 mi^2, lies within the boundaries of two of these blocks; therefore, the 1980-85 data refer to bird species identified over a two-block (21 mi^2) area that contains a second major wetland, the Thousand Acre Swamp, about 3.5 mi east of the Ellison Park wetland. The 1980-85 survey reported 95 species of birds and classified them as possible, probable, or confirmed breeders. Those species that are known to breed in or near a wetland environment, and those that were identified in the 1991-92, 1996, or 2002 surveys, are listed in table 19. Of the 25 wetland species identified in the 1980-85 survey, most were probably sighted in the Ellison Park wetland, although a few species, such as the great blue heron, would be considered visitors to the Ellison Park wetland and breeders in the Thousand Acre Swamp (McKinney, 1996).

The 1991-92 survey (Miller and Ringler, 1992) identified 30 species of birds by sight or call but did not note the breeding status of these species (table 19). Of these species, 23 were identified during the 1980-85 statewide breeding survey. Three species sighted in 1991-92 (pied-billed grebe, herring gull, and osprey) were not identified during any of the other bird surveys.

The 1996 survey (McKinney, 1996) identified 60 species by sight or call (table 19); twice as many as during the 1991-92 survey. Several common species were notably absent (such as sora, American bittern, and least bittern), but the high water levels that year might have hindered breeding activities by these species (McKinney, 1996). Two wetland species that had been found in the past—black tern and blue-winged teal—have not been seen in the area for many years. Two other species—American black duck and sharp-shinned hawk—were not identified during any of the other surveys.

The 2002 survey (Spahn, 2002) was similar in scope and detail to that of McKinney (1996) and reported 91 species (table 19). Thirty-five of these species, of which 19 are potential breeding species, were not detected during the 1996 survey, and 20 were not identified during any other survey. The newly reported species included mute swan (the only confirmed breeder), yellow-headed blackbird (locally very rare), and many warblers. Of the wetland ground-nesting species, no evidence of American bittern, American coot, common moorhen, or pied-billed grebe—which had been reported during at least one of the earlier surveys—was found; least bittern and sora were reported, however. Evidence of cerulean warbler was expected, but this species’ absence is general to the area and not connected to any changes in the wetland (Spahn, 2002).

The composition of the breeding-bird population during 1996 (precontrol period) differed from that in 2002 (postcontrol period) in that 15 additional breeders were identified in 2002; all breeding species reported in 1996 were also identified in 2002. Several ground-nesting species—ring-necked pheasant, common snipe, blue-winged teal, and American woodcock—were not reported during any of the study-period surveys, but only during the 1980-85 survey, which covered an

Table 19. Bird species observed in or near Ellison Park wetland, Monroe County, N.Y., 1980-2002, and designation of breeding status in wetland (possible, probable, or confirmed)

[Based on available data and criteria in Andrle and Carroll (1988). PO, possible breeder; PR, probable breeder; CO, confirmed breeder. X, bird sighted, breeding status unknown. Z, bird sighted but not considered an Ellison Park wetland breeder.]

Common name	Scientific name	Year of survey 1980-85 [a]	1991-92 [b]	1996 [c]	2002 [d]	Ground-nesting species [e]
Grebe, pied-billed	*Podilymbus podiceps*		X			Yes
Heron, black-crowned night	*Nycticorax nycticorax*				Z	
great blue	*Ardea herodias*	PO	X	Z	Z	
green-backed (green)	*Butorides striatus*	PR	X	Z	PO	
Bittern, American	*Botaurus lentiginosus*	PO	X			Yes
Bittern, least	*Ixobrychus exilis*	PR			PO	Yes
Swan, mute	*Cygnus olor*				CO	Yes
Goose, Canada	*Branta canadensis*		X	PO	CO	Yes
Mallard	*Anas platyrhynchos*	CO	X	CO	CO	Yes
Teal, blue-winged	*Anas discors*	CO				Yes
Duck, American black	*Anas rubripes*			Z		
Duck, wood	*Aix sponsa*	CO		CO	CO	
Rail, Virginia	*Rallus limicola*	CO		PR	PO	Yes
Sora	*Porzano carolina*	PR			PO	Yes
Moorhen, common	*Gallinula chloropus*	CO		CO		Yes
Coot, American	*Fulica americana*		X	Z		Yes
Killdeer	*Charadrius vociferus*	CO	X		CO	Yes
Sandpiper, solitary	*Tringa solitaria*				Z	
spotted	*Actitis macularia*	PR	X	PO	PO	Yes
Snipe, common	*Gallinago gallinago*	CO				Yes
Woodcock, American	*Scolopax minor*	CO				Yes
Gull, ring-billed	*Larus delawarensis*		X		Z	
herring	*Larus argentatus*		X			
Tern, caspian	*Sterna caspia*				Z	
Vulture, turkey	*Cathartes aura*			Z	Z	
Hawk, red-tailed	*Buteo jamaicensis*	PO	X	PO	CO	
sharp-shinned	*Accipiter striatus*			Z		
Osprey	*Pandion haliaetus*		X			
Pheasant, ring-necked	*Phasianus colchicus*	CO				Yes
Turkey, wild	*Meleagris gallopavo*				PO	
Dove, mourning	*Zenaida macroura*	CO		PR	CO	
rock	*Columba livia*	CO		CO	PO	
Owl, great-horned	*Bubo virginianus*	CO			PO	
eastern screech	*Otus asio*	CO				
Swift, chimney	*Chaetura pelagica*	PO			Z	
Hummingbird, ruby-throated	*Archilochus colubris*	PR		PR	PO	
Kingfisher, belted	*Ceryle alcyon*	CO	X	PR	CO	
Flicker, northern	*Colaptes auratus*	CO		PR	PO	
Woodpecker, downy	*Picoides pubescens*	CO		CO	CO	
hairy	*Picoides villosus*	CO		CO	CO	
pileated	*Dryocopus pileatus*	CO			PO	
red-bellied	*Melanerpes carolinus*	CO	X	CO	PR	
Kingbird, eastern	*Tyrannus tyrannus*	CO	X	CO	PO	
Flycatcher, alder	*Empidonax alnorum*	PO				
great crested	*Myiarchus crinitus*	CO		PR	CO	
least	*Empidonax minimus*	PO			PO	
willow	*Empidonax traillii*	CO		PO	PR	
Pewee, eastern wood-	*Contopus virens*	CO		PR	CO	
Phoebe, eastern	*Sayornis phoebe*	CO		CO	CO	
Swallow, bank	*Riparia riparia*	CO		CO	PO	
barn	*Hirundo rustica*	CO	X	CO	PO	
cliff	*Hirundo pyrrhonota*				Z	

[a] Includes species identified during 1990-2002 surveys, as well as species identified during New York State Department of Environmental Conservation and Federation of New York State Bird Clubs Breeding Bird Atlas Project (Andrle and Carroll, 1988) that are likely to use a wetland environment for breeding purposes (Robert Spahn, Genesee Ornithological Association, written commun., 1995).

[b] Miller and Ringler (1992) [c] McKinney (1996) [d] Spahn (2002)

[e] Wetland species that nest on, near, or above water surface or on the ground.

Table 19. Bird species observed in or near Ellison Park wetland, Monroe County, N.Y., 1980-2002, and designation of breeding status in wetland (possible, probable, or confirmed)—continued

[Based on available data and criteria in Andrle and Carroll (1988). PO, possible breeder; PR, probable breeder; CO, confirmed breeder. X, bird sighted, breeding status unknown. Z, bird sighted but not considered an Ellison Park wetland breeder.]

Common name	Scientific name	Year of survey				Ground-nesting species [e]
		1980-85 [a]	1991-92 [b]	1996 [c]	2002 [d]	
Swallow, northern rough-winged	*Stelgidopteryx serripennis*			CO	PO	
tree	*Tachycineta bicolor*	CO	X	PR	PO	
Jay, blue	*Cyanocitta cristata*	CO	X	PO	CO	
Crow, American	*Corvus brachyrhynchos*	CO		CO	PR	
Titmouse, tufted	*Parus bicolor*	CO		CO	CO	
Chickadee, black-capped	*Parus atricapillus*	CO	X	CO	CO	
Nuthatch, white-breasted	*Sitta carolinensis*	CO		PR	PO	
Wren, Carolina	*Thryothorus ludovicianus*	CO				
house	*Troglodytes aedon*	CO		CO	PR	
marsh	*Cistothorus palustrus*	CO	X	CO	CO	
Gnatcatcher, blue-gray	*Polioptila caerulea*	CO			CO	
Bluebird, eastern	*Sialia sialis*	PR			PO	
Thrush, wood	*Hylocichla mustelina*	CO		PR	PR	
Veery	*Catharus fuscescens*	CO		PR	PR	Yes
Robin, American	*Turdus migratorius*	CO	X	CO	CO	
Catbird, gray	*Dumetella carolinensis*	CO	X	PR	PR	
Mockingbird, northern	*Mimus polyglottos*		X		PR	
Waxwing, cedar	*Bombycilla cedrorum*	CO	X	PO	PO	
Starling, European	*Sturnus vulgaris*	CO		PO	CO	
Vireo, red-eyed	*Vireo olivaceus*	CO		PO	PR	
warbling	*Vireo gilvus*	CO		PR	PR	
yellow-throated	*Vireo flavifrons*	PR		PR	PR	
Warbler, bay-breasted	*Dendroica castanea*				Z	
blackburnian	*Dendroica fusca*				Z	
black-throated blue	*Dendroica caerulescens*				Z	
black-throated green	*Dendroica virens*				Z	
blue-winged	*Vermivora pinus*	PR			PO	
Cape May	*Dendroica tigrina*				Z	
cerulean	*Dendroica cerulea*	CO				
chestnut-sided	*Dendroica pensylvanica*				Z	
magnolia	*Dendroica magnolia*				Z	
Nashville	*Vermivora ruficapilla*				Z	
Tennessee	*Vermivora peregrina*				Z	
yellow	*Dendroica petechia*	CO		PR	CO	
yellow-rumped	*Dendroica coronata*				Z	
Parula, northern	*Parula americana*	PO			Z	
Yellowthroat, common	*Geothlypis trichas*	CO		PR	PR	
Redstart, American	*Setophaga ruticilla*	CO			PR	
Grosbeak, rose-breasted	*Pheucticus ludovicianus*	CO		PR	CO	
Cardinal, northern	*Cardinalis cardinalis*	CO	X	CO	CO	
Bunting, indigo	*Passerina cyanea*	PR			PO	
Sparrow, chipping	*Spizella passerina*	CO			CO	
house	*Passer domesticus*	CO		CO	CO	
song	*Melospiza melodia*	CO	X	CO	CO	Yes
swamp	*Melospiza georgiana*	CO		PR	PR	Yes
white-crowned	*Zonotrichia leucophrys*				Z	
white-throated	*Zonotrichia albicollis*				Z	
Blackbird, red-winged	*Agelaius phoeniceus*	CO	X	CO	CO	Yes
yellow-headed	*Xanthocephalus xanthocephalus*				Z	
Cowbird, brown-headed	*Molothrus ater*	CO		CO	PR	
Grackle, common	*Quiscalus quiscula*	CO		CO	CO	
Oriole, northern (Baltimore)	*Icterus galbula*	CO	X	CO	CO	
Tanager, scarlet	*Piranga olivacea*	CO		PR	PR	
Goldfinch, American	*Carduelis tristis*	CO	X	PO	PR	
Finch, house	*Carpodacus mexicanus*	CO		CO	PR	
Total number of species		**77**	**30**	**60**	**91**	**20**

area much larger than just the Ellison Park wetland. The absence or scarcity of certain marsh species that were present before or during the early 1980's, such as American bittern, pied-billed grebe, and blue-winged teal, appeared to be regional (Robert Spahn, Genesee Ornithological Association, written commun., 2002), and not associated with the FCS.

Sediment Characteristics

The composition of channel-bottom material within the Ellison Park wetland varies according to flow velocity within a given area. The predominant bed material in the main channel (Irondequoit Creek), in which flow is rapid, was sand; from 82 to 98 percent of the bed material sampled was sand sized or larger (Coon, 1997). The predominant bed materials in the open, backwater areas, where velocities are slow, were silt and clay; from 60 to 96 percent of the bed material was within these particle-size classes (table 20; Coon, 1997). The sediments in these backwater areas contained only small amounts of organic carbon (2 to 7 percent by weight), probably reflecting the relatively slow accumulation of biomass from small submergent aquatic macrophytes (such as *Myriophyllum sp.* and *Ceratophyllum demersum*). Substrate samples from the cattail-covered areas contained much more organic matter (7 to 79 percent by weight), as a result of the high biomass-accumulation rates in these areas (Hunter and others, 2000; Noll and others, 2002).

The FCS was expected to increase the accumulation of sediment and the concentrations of associated metals and organic compounds upstream from the FCS as a result of increased dispersal and detention of stormflows. Sedimentation rates were monitored, and chemical analyses of sediment samples were performed, to assess the effects of the FCS on sediment deposited in the wetland areas upstream and downstream of the FCS.

Sedimentation Rates

Plastic sedimentation disks were anchored to the cattail mat in the wetland at randomly selected locations (fig. 8). Sediment thickness above each disk was measured at least annually, and sedimentation rates were calculated. Sediment was found to be deposited and removed from the disks in a pattern that would be expected for a system in a state of dynamic equilibrium—episodic deposition of a large quantity of sediment in one year, followed in subsequent years by partial resuspension and removal. The sediment included decomposing organic matter, primarily cattail shoots and duckweed; therefore, measurements were adjusted to reflect the inorganic fraction. Errors probably resulted from this adjustment, but the same method was used upstream and downstream from the FCS throughout the study to ensure comparability of data. Therefore, the calculated differences between pre- and postcontrol sedimentation rates are assumed to be valid.

Net sediment accumulation was measured at all sites; 11 of the 12 sites (except SR13; fig. 8) showed a decrease in sediment thickness for at least one annual measurement (table 21). The results from the six long-term measurement sites (SR1 through SR6; fig. 8) indicated that the mean precontrol sedimentation rate for three sites upstream from the FCS was 3.0 mm/yr, and the mean for three sites downstream from the FCS was 2.7 mm/yr. These rates agreed closely with (1) the historic rate of 3 mm/yr computed by Young (1992) from radiocarbon dating of organic matter extracted from two sediment cores from the outlet of Irondequoit Bay and one from the vicinity of Irondequoit Creek near Browncroft Boulevard; (2) the glacial isostatic-rebound rate of 1 ft per century (3 mm/yr) that has been estimated for the south-central shore of Lake Ontario (Clark and Persoage, 1970; Larsen, 1985); and (3) the near-recent rate of 2.8 to 3.7 mm/yr calculated by Schroeder (1985) from lead-210 and cesium-137 radioisotopic measurements of four bottom-sediment cores taken from Irondequoit Bay in 1980. Mean postcontrol sedimentation rates, based on data from the six long-term measurement sites, increased in both wetland areas—to 7.3 mm/yr upstream from the FCS, and 4.6 mm/yr below it (table 21).

Results from the six short-term measurement sites (SR7 through SR10, SR12, and SR13; fig. 8) indicated an inconsistent pattern of postcontrol sediment deposition in the wetland; sedimentation rates at these sites ranged from 3.0 to 66.8 mm/yr. The highest postcontrol sedimentation rate (66.8 mm/yr), which was recorded at SR13, probably reflected (1) this site's location just downstream from where the Millrace channel enters the wetland proper, and the sharp decrease in streamflow velocity as the channelized flows move into this low-gradient, cattail-covered area; and (2) increased sediment loads as a result of the 1997-98 channel modifications that increased the diversion of bankfull stormflows from the creek to the Millrace and caused erosion of the Millrace as the channel adjusted to the increased flows (as identified by comparison of 1992 and 2003 cross-sectional data). No site-to-site trend in sedimentation rates was indicated downstream from SR13; although the rate at the next site downstream (SR12) was 8.0 mm/yr, the next highest sedimentation rate (27.0 mm/yr) was recorded at site SR10 (fig. 8), and rates in the intervening area ranged from 3.4 to 10.0 mm/yr (table 21). The high sedimentation rate at SR10 could reflect the site's proximity to the FCS, which was about 1,000 ft downstream, but the sedimentation rate at SR3, which was between SR10 and the FCS, was 10.6 mm/yr, less than at SR10.

The sedimentation-rate data indicate that the FCS strongly affected sediment deposition in the wetland, as expected, but the annual fluctuations in sediment thickness at any given site were puzzling. The sedimentation rates at upstream and downstream sites are plotted in figure 11 with peak flows in Irondequoit Creek, before and after the installation of the FCS. Note that (1) data from site SR13 (upstream from the FCS) are excluded to make the scale of the two plots and the detail of the data comparable; and (2) four of the nine peak flows that were recorded during the postcontrol period exceeded all peak flows

Table 20. Size distribution of sediment samples collected for chemical analyses from Ellison Park wetland, Monroe County, N.Y., 1994-2001, by percentage of particles finer than index-particle size

[SQ, sediment-quality site. US (DS), upstream (downstream) from flow-control structure. Locations are shown in fig. 8.]

Sampling site and wetland area	Date of sample collection	Index particle size, in millimeters: 2	1	0.5	0.25	0.125	0.062	0.031	0.016	0.008	0.004	0.002	0.001	Percentage of sample that falls in given particle-size class: Sand	Silt	Clay
		Percentage of particles finer than particle size given above														
SQ1 - US	10/1994	100	99.99	99.96	99.84	99.28	92.39	89.04	62.58	37.11	21.62	11.55		7.6	70.8	21.6
	10/1997				100	99.8	89.0	69.39	43.59	29.80	19.57	11.56	6.23	11.0	69.4	19.6
	11/1999				100	98.4	94.5	65.97	43.57	29.96	19.00	9.92	6.99	5.5	75.5	19.0
	10/2001			100	99.8	99.2	94.9	78.8	55.8	38.7	35.1	27.0		5.1	59.8	35.1
SQ2 - US	10/1994		100	99.65	99.16	97.35	88.69	76.46	54.03	34.23	24.76	16.46		11.3	63.9	24.8
	10/1997			100	98.7	93.6	82.4	74.18	57.70	43.38	29.50	18.22	12.15	17.6	52.9	29.5
	11/1999		100	98.2	94.8	87.0	81.9	59.8	30.3	21.1	13.3	7.3	7.1	18.1	68.6	13.3
	10/2001				100	98.9	95.5	82.6	66.0	41.5	27.0	23.6		4.5	68.5	27.0
SQ3 - DS	10/1994	100	99.93	99.77	99.38	97.87	82.35	35.47	32.41	19.93	14.80	10.91		17.6	67.6	14.8
	10/1997				100	99.4	79.2	57.39	36.02	26.52	18.21	11.87	8.31	20.8	61.0	18.2
	11/1999			100	93.50	89.3	59.5	36.50	24.97	15.64	8.98	4.58	3.03	40.5	50.5	9.0
	10/2001				100	95.2	67.7	35.2	22.4	17.1	15.1	12.0		32.3	52.6	15.1

Table 21. Sediment thickness and net sedimentation rates in Ellison Park wetland, Monroe County, N.Y., 1991-2001

[SR, sedimentation-rate measurement site. mm/yr, millimeter per year; --, no data; US (DS), upstream (downstream) from flow-control structure. Locations are shown in fig. 8.]

Sampling site and wetland area	Annual thickness of accumulated sediment, in millimeters: 1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	Precontrol period (1991-96): Net sediment thickness (mm)	Period of data collection (years)	Net sedimentation rate (mm/yr)	Postcontrol period (1997-2001): Net sediment thickness (mm)	Period of data collection (years)	Net sedimentation rate (mm/yr)
Long-term sites																		
SR1 - US	0	2	16	9	6	6	30	48	33	37	43	47	6	5	1.2	17	5	3.4
SR2 - US	0	4	15	--	6	8	--	--	--	20	40	--	8	5	1.6	32[a]	4	8.0
SR3 - US	0	17	16	12	15	31	50	63	66	86	91	103	31	5	6.2	53	5	10.6
SR4 - DS	0	20	11	15	--	11	79	99	97	106	91	99	11	5	2.2	20	5	4.0
SR5 - DS			0	30	--	9	--	--	21	35	30	39	9	3	3.0	30[a]	5	6.0
SR6 - DS			0	9	--	--	--	38	49	33	24	28	9[a]	3	3.0	19[a]	5	3.8
Short-term sites																		
SR7 - US					0	2	4	38	36	44	44	--				40	4	10.0
SR8 - DS							0	26	29	23	30	36				36	5	7.2
SR9 - DS							0	7	--	4	27	15				15	5	3.0
SR10 - US							0	18	14	13	98	135				135	5	27.0
SR12 - US							0	0	2.5	28	27	40				40	5	8.0
SR13 - US							0	105	140	256	278	334				334	5	66.8

[a] Sediment thickness estimated.

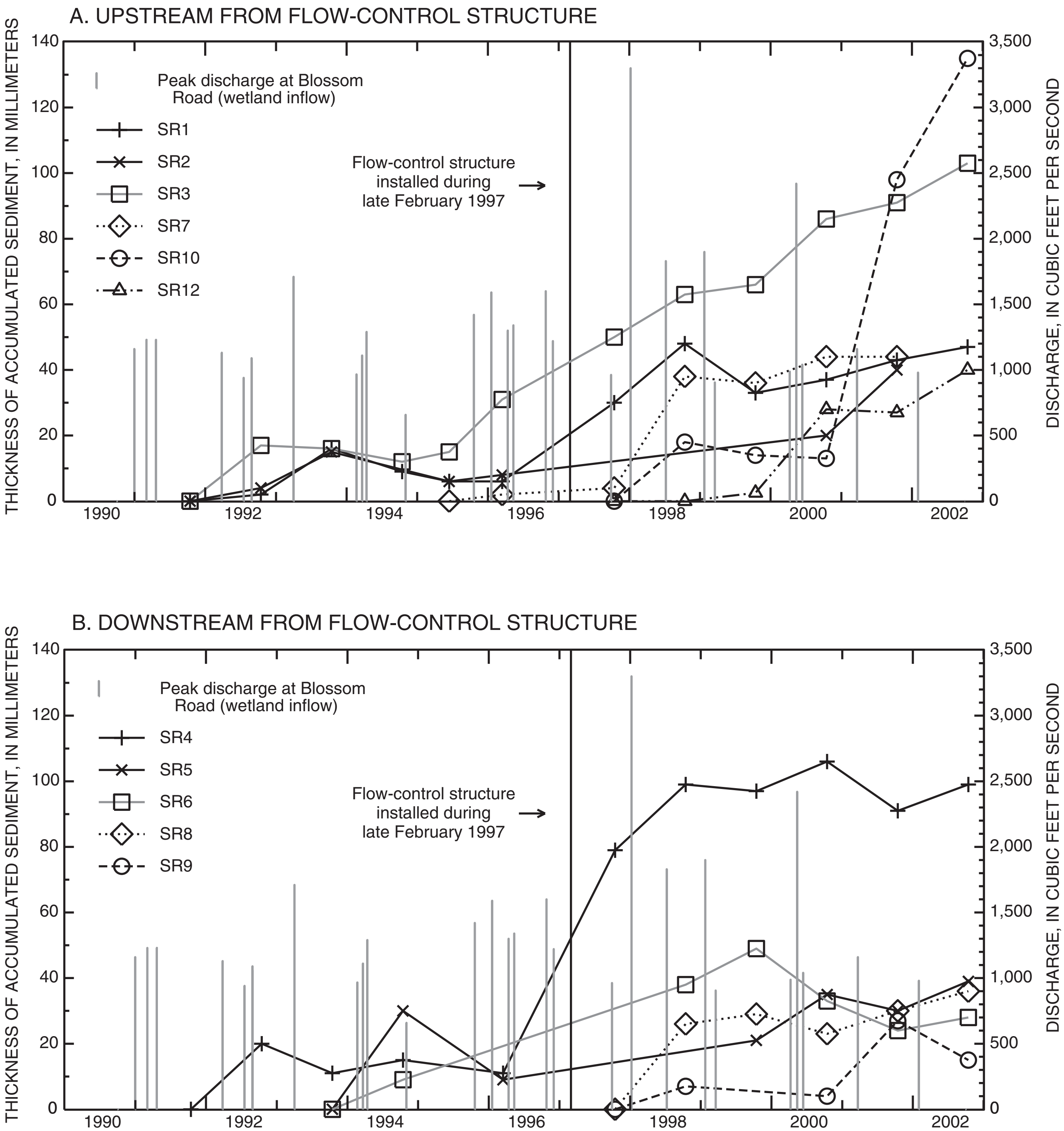


Figure 11. Thickness of accumulated sediment at sedimentation-rate-measurement sites in Ellison Park wetland, Monroe County, N.Y., 1991-2002: A. Upstream from flow control structure. B. Downstream from flow-control structure.

that occurred during the precontrol study period (1991-96) and the previous 10-year period (1981-90). Sediment thickness at all long-term sites (SR1 through 6) increased during 1996-98; the FCS was installed during 1997. The plotted data indicate that (1) most sites showed an increase in sediment accumulation that began in 1995 or 1996, before installation of the FCS; (2) the greatest increase occurred at SR4, downstream from the FCS; (3) many of the sites indicated a leveling off of sediment accumulation both upstream and downstream from the FCS after 1998; and (4) despite the increased frequency of high flows during the postcontrol period, no relation between the magnitude of peak flows and the annual change in sediment thickness was found.

High peakflows did not necessarily result in greater sediment deposition in the wetland than low peak flows. Peak flows that coincided with seasonally high water levels in the wetland (fig. 4) might explain more of the variation in sediment accumulation than peak-flow magnitude alone. At most sites, sediment accumulation during high-water periods was greater than during low-water periods, regardless of the peak-flow magnitude, although even this relation was inconsistent. Therefore, although the FCS undoubtedly had an effect, the apparent increase in sedimentation rates after FCS installation could have been partly the result of other factors, such as the magnitude, frequency, and the timing (season) of peak flows.

Sediment Quality

Sediment samples were collected periodically from three sites in the wetland (fig. 8) and analyzed for major and trace elements (table 22A), polycyclic aromatic hydrocarbons (PAHs; table 22B), and organochlorine and organophosphate compounds (table 22C). The percentage of silt-plus-clay in the sediment can affect the concentrations of metals (Hem, 1985), and the percentage of organic matter can affect the concentrations of organic compounds (Smith and others, 1988). The percentage of fine-grained sediment in a sample, or the concentration of aluminum or titanium, can be used to normalize trace-element concentrations for identification of trends and adjustment for manmade effects, and to make data comparable among sites and among years (Horowitz, 1991); similarly, total organic carbon concentrations can be used to normalize concentrations of organic compounds (Daskalakis and O'Connor, 1995). Normalization of the Ellison Park wetland samples was unnecessary because (excluding the 1999 sample from site SQ3, which was anomalous as explained below) (1) more than 68 percent of each sample was silt and clay size (table 20); (2) aluminum concentrations varied by only 0.7 percent among the 11 samples, and titanium concentrations by only 0.03 percent; and (3) total organic carbon concentrations varied by only 3.6 percent among the 11 samples (table 22A).

The 1999 sample from site SQ3 was anomalous in that it contained many trace elements in unusually high concentrations and many major elements in unusually low concentrations in relation to the concentrations in samples from other sites and other years. This sample also had the smallest amount of silt and clay and the lowest organic-carbon concentration of all samples collected, but no other distinction was identified that could explain the discrepancies in element concentrations. The results from this sample are included in table 22A but were not used in the element analyses.

The high concentrations of some major elements, including barium, manganese, strontium, and zinc, and some trace elements, including chromium, copper, lead, and vanadium (table 22A), were within the range expected for sediment derived from an urbanized basin (Coon and others, 2000). The concentrations of most PAHs (coal-tar compounds) were below the analytical detection limit and are not included in table 22B. PAHs that consistently had measurable concentrations were benzo[a] anthracene, benzo[a] pyrene, benzo[b] and benzo[k] fluoranthene, benzo[g,h,i] perylene, bis(2-ethylhexyl) phthalate, chrysene, fluoranthene, phenanthrene, and pyrene (table 22B). Organochlorine compounds, including pesticides and polychlorinated biphenyls, were found at measurable levels, whereas organophosphate compounds (pesticides) generally were not (table 22C). The concentrations of many of the organochlorine compounds were comparable to or lower than those measured in 14 studies in the United States during the 1970's and 1980's, as reported by Smith and others (1988). Many of the organic compounds found in the sediment samples are (1) highly resistant to chemical or biological transformation and, therefore, extremely persistent in the environment; and (2) have low aqueous solubility and strongly partition from solution to biota, particulate material, and dissolved organic matter (Smith and others, 1988). Therefore, the presence and concentrations of these compounds were fairly typical of a depositional environment in a highly urbanized basin.

Sediment-quality guidelines (SQGs) have been developed to assess sediment quality and to set targets for sediment quality that will sustain ecosystem health in aquatic environments (Canadian Council of Ministers of the Environment, 1995; Ingersoll and others, 2000). SQGs identify threshold concentrations of selected chemicals that, if exceeded in freshwater or marine sediments, are likely to cause adverse effects in aquatic life. Three freshwater SQGs for 7 metals, 11 PAHs, and 9 organochlorine compounds are presented in table 22. The interim sediment-quality guideline (ISQG) is the concentration below which adverse effects to aquatic life are not expected to occur; the probable effect level (PEL) is the concentration above which adverse effects to aquatic life are expected to frequently occur (Canadian Council of Ministers of the Environment, 1995). The consensus-based probable effect concentration (PEC), which is based on agreement among various published SQGs (MacDonald and others, 2000), are alternative threshold concentrations above which adverse effects are expected to frequently occur (Ingersoll and others, 2000). PEC values are generally greater than PEL values. Comparison of the measured concentrations of selected elements and compounds in the wetland sediment samples and

Table 22. Chemical concentrations in sediment samples from Ellison Park wetland, Monroe County, N.Y., 1994-2001, and sediment-quality guidelines for selected chemicals

[SQ, sediment-quality site. m, presence verified, but not quantified; e, estimated; <, less than. Site locations are shown in fig. 8.]

Element	Freshwater sediment quality guideline			Southern wetland area (upstream from flow-control structure)								Northern wetland area (downstream from flow-control structure)			
				Site SQ1				Site SQ2				Site SQ3			
	ISQG[1]	PEL[2]	PEC[3]	1994	1997	1999	2001	1994	1997	1999	2001	1994	1997	1999	2001
A. Major and trace elements, values represent concentrations in clay fraction (particle size < 63 microns), in micrograms per gram (parts per million) unless indicated as percentage (%) of sample by weight															
Aluminum, %				5	5	4.9	4.3	4.9	5	4.7	4.4	4.8	4.9	9.8	4.3
Antimony				0.4	0.4	0.4	0.5	0.2	0.4	0.4	0.7	0.6	0.4	1	0.3
Arsenic	5.9	17.0	33.0	5	3.4	4.2	6	5	3.7	4.6	4.3	4	3.4	15	2.8
Barium				460	430	410	410	450	430	400	400	470	450	560	440
Beryllium				1	<1	1.2	1.1	1	1	1.1	1.3	1	1	2.6	0.9
Bismuth				<10	<10	<1	<1	<10	<10	<1	<1	<10	<10	<1	<1
Cadmium	0.6	3.5	4.98	0.8	0.6	0.8	0.8	0.9	0.7	0.8	0.8	3.8	2.6	0.4	1.9
Calcium, %				6.5	6.5	6.4	7	6.7	6.5	8.1	8.9	4.7	4.5	2.1	4.7
Cerium				44	49	46	41	45	48	46	44	39	44	73	36
Chromium	37.3	90.0	111	43	38	43	53	46	44	43	52	59	45	85	50
Cobalt				10	11	7	8	12	7	8	8	10	6	13	6
Copper	35.7	197	149	40	32	41	41	49	37	41	43	72	43	35	35
Europium				<2	<2	1	<1	<2	<2	1	<1	<2	<2	1	<1
Gallium				12	16	10	9	12	14	10	10	11	13	22	9
Gold				<8	<8	<1	<1	<8	<8	<1	<1	<8	<8	<1	<1
Holmium				<4	<4	<1	<1	<4	<4	<1	<1	<4	<4	<1	<1
Iron, %				2.7	2.3	2.5	2.5	2.8	2.7	2.6	2.5	2.4	2.2	4.8	1.9
Lanthanum				25	25	24	19	25	27	24	21	23	24	42	17
Lead	35.0	91.3	128	45	42	49	52	62	48	49	50	89	53	28	40
Lithium				30	20	30	20	30	30	30	30	20	20	60	20
Magnesium, %				1.4	1.2	1.3	1.2	1.2	1	1.1	1.2	1.3	1.1	1.4	1
Manganese				880	630	520	590	720	640	630	670	530	590	860	430

[1] Interim freshwater sediment quality guideline; concentration below which adverse effects to aquatic life are not expected to occur (Canadian Council of Ministers of the Environment, 1999).

[2] Probable effect level; concentration above which adverse effects to aquatic life are expected to frequently occur (Canadian Council of Ministers of the Environment, 1999).

[3] Consensus-based probable effect concentration; concentration above which adverse effects to aquatic life are expected to frequently occur (Ingersoll and others, 2000).

Table 22. Chemical concentrations in sediment samples from Ellison Park wetland, Monroe County, N.Y., 1994-2001, and sediment-quality guidelines for selected chemicals—continued

Element	Freshwater sediment quality guideline			Southern wetland area (upstream from flow-control structure)								Northern wetland area (downstream from flow-control structure)			
				Site SQ1				Site SQ2				Site SQ3			
	ISQG[1]	PEL[2]	PEC[3]	1994	1997	1999	2001	1994	1997	1999	2001	1994	1997	1999	2001
Mercury	0.17	0.486	1.06	0.1	0.12	0.14	0.16	0.14	0.1	0.13	0.16	0.33	0.19	0.03	0.16
Molybdenum				<2	<2	0.5	0.6	<2	<2	0.5	0.6	<2	<2	0.9	<0.5
Neodymium				26	25	22	22	24	25	22	22	23	24	33	19
Nickel	--	36	48.6	18	15	20	18	20	19	21	19	20	16	40	13
Niobium				12	9	8	8	11	10	7	8	11	9	17	7
Phosphorus, %				0.13	0.12	0.13	0.13	0.15	0.13	0.13	0.14	0.12	0.11	0.099	0.087
Potassium, %				1.7	1.7	1.6	1.4	1.6	1.6	1.6	1.5	1.6	1.7	2.2	1.5
Scandium				8	7	8	8	8	8	8	8	7	7	16	6
Selenium				0.6	0.5	0.6	0.8	0.8	0.6	0.6	0.8	0.8	0.4	0.9	0.4
Silver				0.5	0.4	0.4	0.3	0.7	0.6	0.4	0.7	2	0.9	0.7	0.5
Sodium, %				1.2	1.2	1.1	0.84	1.1	0.95	0.99	0.79	1.3	1.3	0.25	1.2
Strontium				420	390	360	450	440	440	470	550	340	320	130	350
Sulfur, %				0.34	0.63	0.64	0.81	0.6	0.54	0.49	0.68	0.35	0.26	<0.05	0.28
Tantalum				<40	<40	<1	<1	<40	<40	<1	<1	<40	<40	2	<1
Thallium						<1	<1			<1	<1			<1	<1
Thorium				6	5	5	5	6	5	5	5	6	5	15	4
Tin				<5	<5	4	4	<5	<5	4	4	<5	<5	4	4
Titanium, %				0.26	0.25	0.26	0.24	0.25	0.27	0.25	0.24	0.27	0.25	0.39	0.26
Uranium				1.6	1.5	1.4	1.2	1.6	1.7	1.3	1.2	1.7	1.5	2.7	1
Vanadium				50	46	47	47	50	51	48	48	45	44	150	36
Ytterbium				2	2	2	2	2	2	2	2	2	2	2	2
Yttrium				20	22	22	20	19	22	21	19	19	21	30	18
Zinc	123	315	459	220	170	200	240	230	190	200	250	250	200	140	170
Total carbon, %				5.4	6.6	6.7	8.9	6.9	7.3	6.5	7.6	5.1	4.6	2	3.8
Inorganic carbon, %				1.9	1.8	1.7	2.1	1.9	1.8	2.1	2.5	1.4	1.3	0.33	1.2
Organic carbon, %				3.5	4.8	5	6.9	5	5.5	4.4	5.2	3.7	3.3	1.6	3.7

Table 22. Chemical concentrations in sediment samples from Ellison Park wetland, Monroe County, N.Y., 1994-2001, and sediment-quality guidelines for selected chemicals—continued

Element	Freshwater sediment quality guideline			Southern wetland area (upstream from flow-control structure)								Northern wetland area (downstream from flow-control structure)			
				Site SQ1				Site SQ2				Site SQ3			
	ISQG[1]	PEL[2]	PEC[3]	1994	1997	1999	2001	1994	1997	1999	2001	1994	1997	1999	2001
B. Polycyclic aromatic hydrocarbons, values represent concentrations in silt-clay fraction (particle size < 2 mm), in micrograms per kilogram (parts per billion)															
1,2,5,6-Dibenz-anthracene				<400	<400	m	--	<400	<400	m	--	<400	<400	e100	--
Acenaphthene	6.71	88.9	--	<200	<200	m	e10	<200	--	m	<150	<200	--	m	e30
Acenaphthyl-ene	5.87	128	--	<200	<200	m	e20	<200	--	m	e30	<200	--	m	e50
Anthracene	46.9	245	845	<200	--	m	e100	<200	--	m	e110	<200	--	e100	140
Benzo[a] anthracene	31.7	385	1,050	500	--	e200	500	<400	--	e200	560	1,200	500	400	590
Benzo[a]pyrene	31.9	782	1,450	600	--	e300	700	500	--	e300	750	1,300	600	600	720
Benzo[b] fluoranthene				1,000	--	600	1,000	800	500	700	1,100	1,300	1,000	1,000	1,200
Benzo[k] fluoranthene				1,000	--	e200	900	700	500	e200	820	1,400	600	e400	740
Benzo[g,h,i] perylene				<400	<400	e200	470	<400	--	e200	540	800	--	e400	480
Bis(2-ethylhexyl) phthalate				500	--	e200	--	500	--	e200	--	800	300	400	--
Chrysene	57.1	862	1,290	900	--	500	1,000	700	--	500	1,000	1,700	700	800	960
Fluoranthene	111	2,355	2,230	1,800	300	700	2,000	1,100	500	700	1,900	3,300	1,600	1,000	1,700
Fluorene	21.2	144	536	<200	<200	m	--	<200	--	m	--	<200	--	m	--
Indeno(1,2, 3-cd) pyrene				<400	<400	e200	580	<400	--	e200	700	700	--	e400	640
Isophorone				<200	<200	<200	<150	<200	--	m	<150	<200	<200	m	<100
Naphthalene	34.6	391	561	<200	<200	m	e20	<200	--	m	e20	<200	--	m	e30
n-Butylbenzyl-phthalate				<200	<200	m	--	<200	<200	m	--	<200	<200	m	--
Phenanthrene	41.9	515	1,170	700		200	590	300		e200	460	1,700	500	400	610
Phenol				<200	<200	<200	e80	<200	<200	<200	e90	<200	--	<200	e40
Pyrene	53.0	875	1,520	1,400	300	600	1,500	800	400	600	1,500	2,900	1,200	900	1,300

Table 22. Chemical concentrations in sediment samples from Ellison Park wetland, Monroe County, N.Y., 1994-2001, and sediment-quality guidelines for selected chemicals—continued

Element	Freshwater sediment quality guideline			Southern wetland area (upstream from flow-control structure)								Northern wetland area (downstream from flow-control structure)			
				Site SQ1				Site SQ2				Site SQ3			
	ISQG[1]	PEL[2]	PEC[3]	1994	1997	1999	2001	1994	1997	1999	2001	1994	1997	1999	2001
C. Organochlorine and organophosphate compounds, values represent concentrations in silt-clay fraction (particle size < 2 mm), in micrograms per kilogram (parts per billion)															
Organochlorine compounds															
Aldrin				<0.1	0.5	<0.2	<2	<0.2	<0.5	<0.8	<3	<0.1	<0.3	<0.8	<1
Chlordane	4.50	8.87	17.6	20	43	24	--	18	40	38	--	26	39	31	--
Dieldrin	2.85	6.67	61.8	2.0	2.2	1.4	m	1.2	1.5	3	<3	3.5	4.2	3.9	m
Endosulfan				<0.1	<0.2	<0.2	<2	<0.2	<0.2	<0.2	<3	<0.1	<0.2	<0.2	<1
Endrin	2.67	62.4	207	<0.8	<1.2	<0.2	<4	<1.6	<1.1	<0.2	<6	<0.8	<0.5	<0.2	<2
Heptachlor				<0.1	<0.2	<0.2	<2	<0.2	<0.2	<0.2	<3	<0.1	<0.2	<0.2	<1
Heptachlor epoxide	0.60	2.74	16.0	<0.8	0.4	<0.4	<2	<1.6	0.4	<0.5	<3	<0.8	<0.2	<0.4	<1
Lindane	0.94	1.38	4.99	<0.1	<0.2	<0.2	<2	<0.2	<0.2	<0.2	<3	<0.1	<0.2	<0.2	<1
Methoxychlor				<29	<2.5	<2.5	<10	<24	<2.5	<2.5	<15	<38	<2.5	<2.5	<5
Mirex				<0.1	<0.2	<0.2	<2	<0.2	<0.2	<0.2	<3	<0.1	<0.2	<0.2	<1
p,p'-DDD	3.54	8.51	28.0	3.4	5.8	e6.7	3	4.5	4.6	e13	e3	6.5	7.8	e11	e7
p,p'-DDE	1.42	6.75	31.3	6.6	13	12	7	8.4	11	12	9	7.4	11	12	8
p,p'-DDT	1.19	4.77	62.9	1	1.4	<2.7	<4	0.6	1.2	<5.0	<6	0.8	1.8	<2.8	e1
PCB	34.1	277	676	26	46	39	<100	43	40	75	<150	52	98	69	60
PCN				<1.0	--	--	--	<2.0	--	--	--	<1.0	--	--	--
Toxaphene	0.1	--	--	<10	<50	<50	--	<20	<50	<50	--	<10	<50	<50	--
Organophosphate compounds															
Diazinon				--	1.8	<0.4	<0.2	--	1.7	0.5	<0.2	--	3	0.4	<0.2
Ethion				--	<0.2	<0.4	<0.2	--	<0.2	<0.2	<0.2	--	<0.2	<0.2	<0.2
Malathion				--	<0.2	<0.4	<0.2	--	<0.2	<0.2	<0.2	--	<0.2	<0.2	<0.2
Methyl parathion				--	<0.2	<0.4	<0.2	--	<0.2	<0.2	<0.2	--	<0.2	<0.2	<0.2
Parathion				--	<0.2	<0.4	<0.2	--	<0.2	<0.2	<0.2	--	<0.2	<0.2	<0.2
Trithion				--	--	<0.2	<0.2	--	--	<0.2	<0.2	--	--	<0.2	<0.2

the SQGs indicates that (1) concentrations of metals exceeded the ISQG values but generally were lower than the PEL and PEC values; this implies that the wetland sediment quality is slightly impaired; (2) concentrations of PAHs greatly exceeded ISGQ values and often exceeded the PEL values, and exceeded the PEC values at Site SQ3 during 1994, which implies severe impairment; and (3) concentrations of several organochlorine compounds were less than the ISQG values, although the concentrations of DDD, DDE, DDT, and PCB indicated moderate impairment, and chlordane concentrations indicated severe impairment. These results generally indicate that the ecological health of the wetland is moderately to severely impaired, and agree with the evaluation of ecological health based on the 1996 macroinvertebrate survey. Except for the elevated concentrations of PAHs at site SQ3 downstream from the FCS in 1994, the data indicated a similar degree of impairment in each part of the wetland during a given year.

Most of the elements and compounds showed no substantial fluctuation in concentrations, nor a consistent upward or downward trend, among the years sampled. The only elements or compounds whose concentrations indicated a clear upward trend from 1994 to 2001 upstream from the FCS were antimony, calcium, chromium, and possibly mercury and strontium (table 22A); the only one of these to show an upward trend downstream from the FCS was chromium. Downward trends were indicated for barium and uranium. Concentrations of many PAHs appeared to decrease in 1997 and 1999, but the 2001 concentrations were close to those measured in 1994 (table 22B). Conversely, concentrations of the few organochlorine compounds that had detectable concentrations in all four years appeared to increase in 1997 and 1999, but like many PAHs, their 2001 concentrations were comparable to those measured in 1994 (table 22C). Therefore, these results do not indicate that the FCS had any definable effect on sediment quality.

Constituent Loads and Removal Efficiencies

Inflow and outflow loads of selected constituents, and the wetland's removal efficiency for each constituent, were computed to evaluate the effect of the FCS on constituent loads. Bias in the concentrations of some constituents collected by the automatic samplers at the inflow and outflow water-quality monitoring sites might affect these results, as discussed below.

Bias in Water-Quality Data

Significant differences in the means of the paired EWI- and Auto-sample data, and indications of systematic bias in the data collected by the automatic sampler at the inflow site (Blossom Road), were noted in the concentrations of total phosphorus, ammonia, and total suspended solids during the pre- and postcontrol periods, and in the concentrations of volatile suspended solids during the postcontrol period. Bias in the automatically collected samples from the outflow site (Empire Boulevard) was noted in the concentrations of ammonia during the pre- and postcontrol periods, and nitrate-plus-nitrite nitrogen during the postcontrol period (table 23). The concentrations of automatically collected samples were not adjusted for these biases before calculation of loads for the following five reasons.

1. The QA/QC analyses were performed on discrete samples that were associated with instantaneous discharges, whereas the automatic-sample analyses were performed on samples composited over periods ranging from several hours to several days and were associated with the mean discharge over that period.
2. The majority of QA/QC samples were collected during low-flow periods, mostly when streamflow was less than 300 ft^3/s, and none were collected during annual peak-flow periods.
3. No consistent relation between the errors in concentration and discharge was observed. The variation in error was generally greatest among low-flow samples and decreased with increasing discharge; therefore, any error in the computed loads presumably was greatest during low-flow periods, when the loads of a constituent in the creek were small, and was smallest during periods when the loads in the creek were large.
4. For a given constituent, the range in concentration of the QA/QC samples collected during either the pre- or postcontrol period was much less than the range encompassed by all of the automatically collected samples that were collected during that period (see table 7).
5. Biases, when identified for both monitoring sites during a given period, such as for ammonia concentrations, had the same sign (+ or -) at both sites.

These observations indicate that error in the computed loads of the five constituents named above was unavoidable, but the relation between inflow and outflow loads, and the relative change in wetland removal efficiency, was valid. The possible effects of using unadjusted concentration data for estimation of chemical loads are discussed in the sections on specific constituents.

Criteria for Evaluation of Flow-Control Structure

The effect of the FCS on constituent concentrations and loads passing through the wetland was assessed through a comparison of pre- and postcontrol data from the wetland's inflow and outflow (1) chemical concentrations (table 24, fig. 12); (2) average annual loads (table 25); and (3) monthly loads (including postcontrol outflow loads that were predicted by the precontrol ESTIMATOR-regression equation) and flow-weighted concentrations (figs. 13-21; appendix 1). These data were used to calculate annual and monthly pre- and postcontrol removal-efficiency values (figs. 13-21; appendix 1). Statistical analyses were performed to objectively identify significant

Table 23. Mean difference (bias) between concentrations of selected constituents in water samples collected by automatic sampler (Auto) and equal-width-increment method (EWI), and comparative statistics for concentrations of constituents in quality-assurance/quality-control samples (QA/QC) and all automatically collected samples at the inflow (Blossom Road) and outflow (Empire Boulevard) of Ellison Park wetland, Monroe County, N.Y., 1990-2001

[Values are in milligrams per liter. Bias is mean difference between concentrations in Auto and EWI samples for those constituents for which a statistically significant difference between concentrations was found. A positive (negative) bias results when Auto concentrations are consistently greater than (less than) EWI concentrations.
n, number of samples; Min, minimum value; Max, maximum value; <, less. than. Locations are shown in fig. 2.]

	Inflow (Blossom Road)									Outflow (Empire Boulevard)								
	QA/QC data					All data				QA/QC data					All data			
Constituent	n	Bias	Median	Min	Max	n	Median	Min	Max	n	Bias	Median	Min	Max	n	Median	Min	Max
Precontrol period (October 1990 through February 1997)																		
Total phosphorus	19	0.035	0.10	0.02	0.37	1081	0.10	0.009	2.15									
Ammonia	18	-.005	.01	< .01	.07	1045	.01	< .01	.32	14	-0.015	0.03	0.01	0.09	925	0.03	0.009	0.66
Total suspended solids	15	32.7	56	6	188	324	139	3	1,510									
Postcontrol period (March 1997 through September 2001)																		
Total phosphorus	17	0.018	0.055	.02	.20	636	0.075	.015	6.75									
Ammonia	15	-.01	0.02	< .01	.04	657	< .01	< .01	.45	17	-0.016	0.06	.01	.09	694	0.03	.01	0.2
Nitrate plus nitrite										17	.034	0.90	.51	1.4	633	0.9	.4	2.6
Total suspended solids	15	9.34	13	7.7	142	195	83	4	5,800									
Volatile suspended solids	11	1.61	3	1.6	14	193	12	2	293									

Table 24. Concentrations and summary statistics of selected constituents, Ellison Park wetland, Monroe County, N.Y., 1990-2001

[Inflow site is Blossom Road; outflow site is Empire Boulevard. Concentrations are in milligrams per liter except those for cadmium, copper, lead, and zinc, which are in micrograms per liter. n, number of samples; <, less than. Locations are shown in fig. 2.]

	Precontrol period (October 1990 through February 1997)								Postcontrol period (March 1997 through September 2001)							
	Inflow concentration				Outflow concentration				Inflow concentration				Outflow concentration			
		Percentile				Percentile				Percentile				Percentile		
Constituent	n	75th	50th	25th	n	75th	50th	25th	n	75th	50th	25th	n	75th	50th	25th
Total phosphorus	1081	0.17	0.10	0.06	1004	0.16	0.10	0.06	636	0.14	0.08	0.04	606	0.14	0.09	0.06
Orthophosphate	1082	.016	.011	.007	947	.023	.016	.01	663	.02	.013	.009	704	.026	.017	.011
Ammonia-plus- organic nitrogen	1076	.99	.77	.58	1003	.98	0.78	.61	633	.65	.46	.33	608	.68	.52	.41
Ammonia nitrogen	1045	.02	.01	< .01	925	.05	.03	.02	657	.015	< .01	< .01	694	.05	.03	.02
Nitrate plus nitrite	1066	1.2	.99	.81	961	1.2	.9	.7	623	1.2	1.0	.8	633	1.1	.9	.7
Total suspended solids	324	244	139	95	255	152	102	73	195	171	83	39	162	116	63	28
Volatile suspended solids	320	30.5	21	15	242	21	16	12	193	21	12	7	162	16	11	6
Chloride	1087	140	110	95	945	140	120	100	626	150	130	110	637	150	130	115
Sulfate	1076	180	140	110	934	170	140	110	627	180	150	110	637	170	140	100
Total organic carbon	22	6.7	6.0	5.0	23	6.3	5.6	4.4	141	8.2	6.1	5.2	147	7.8	6.2	5.3
Biochemical oxygen demand	50	3.0	< 2.0	< 2.0	53	2.2	< 2.0	< 2.0	139	3.0	2.1	< 2.0	145	2.9	2.2	< 2.0
Chemical oxygen demand	24	40	30	18	27	30	22	14	147	40	30	20	153	40	30	20
Cadmium	76	<1	< 1	< 1	78	< 1	< 1	< 1	153	1	1	< 1	159	< 1	< 1	< 1
Copper	70	70	40	20	70	70	40	30	153	60	40	30	159	60	40	30
Lead	74	13	7	< 5	76	8	5	< 5	153	19	11	6	159	16	11	5
Zinc	775	70	40	< 40	652	60	40	< 30	303	40	25	15	304	35	20	15

differences in loads and flow-weighted concentrations between inflow and outflow sites and between the pre- and postcontrol periods (table 26). Included in these analyses was the adjustment of postcontrol loads and flow-weighted concentrations by the mean differences between precontrol inflow and outflow loads and flow-weighted concentrations, respectively, to statistically remove the load-altering effect of the wetland ("wetland effect") and theoretically isolate the effect of the FCS. (See "Statistical Analyses of Load Data" in the "Methods" section.)

Total Phosphorus

Inflow concentrations of total phosphorus (TP) generally were lower during the postcontrol period than the precontrol period (table 24; fig. 12), but the average annual inflow load increased 13 percent (table 25). This increase reflected the disproportionate runoff volume that occurred during 2 of the 5 postcontrol years. Postcontrol outflow concentrations of TP were slightly lower than precontrol outflow concentrations, and average annual outflow loads decreased 14 percent. Outflow loads exceeded inflow loads during the dry summers of 1999 and 2001 but generally were lower than the loads predicted by the precontrol regression equation during these periods; they also were much lower during the postcontrol high-flow months of January and July 1998 and May 2000 than during the precontrol high-flow months of July 1992 and April 1993 (fig. 13). Outflow flow-weighted concentrations generally were lower than inflow flow-weighted concentrations throughout the study period. The wetland's removal efficiency for TP increased from 28 percent during the precontrol period to 45 percent during the postcontrol period (table 25).

The statistical analyses indicate that outflow loads were significantly lower than inflow loads during the precontrol period ($p < 0.0001$); however, the extremely large differences between inflow and outflow loads that resulted from the floods of January and July 1998 and May 2000 increased the variance in the data such that no statistically significant difference between inflow and outflow loads was detected for the postcontrol period ($p = 0.0932$; table 26). By removing the effect of the hydrologic dissimilarity between pre- and postcontrol periods from the load data through an analysis of the flow-weighted concentrations, the outflow concentrations were significantly lower than inflow concentrations for both periods ($p < 0.0001$, $p = 0.0078$). Additionally, adjusting the postcontrol data by removing the precontrol "wetland effect" resulted in a significant improvement in TP load reduction that could be attributed to the FCS; this adjustment decreased the p value for the postcontrol differences in flow-weighted concentrations from 0.0078 to 0.0012.

TP concentrations in the automatically collected samples from the inflow site showed a positive bias during the pre- and postcontrol periods (table 23); that is, the TP concentrations in the Auto (automatically collected) samples generally were higher than the mean concentration of EWI (cross-sectional equal-width-increment) samples. No bias was found in the outflow concentrations. The magnitude of the inflow bias varied greatly in low-flow samples; the difference between Auto and EWI concentrations ranged from 0 to almost 80 percent and averaged about 30 percent. The variation decreased with increasing discharge—the difference between the Auto and EWI concentrations (0.19 and 0.18 mg/L, respectively) for the highest sampled flow (800 ft^3/s) was within the analytical error of measurement. Therefore, the greatest error in the computed loads presumably results from low-flow periods, when the TP load was small, and the smallest error results from high-flow periods, when the TP load was large. If the error was large, the computed inflow load (and the associated removal efficiency) for the precontrol period would have been inordinately large. The average TP bias (0.018 mg/L) during the postcontrol period, was only half as large as the precontrol bias (0.035 mg/L); thus, the error in the computed load and removal efficiency would have been smaller. Therefore, the relation between inflow and outflow loads, and the relative change in removal efficiency, are considered valid, even though the actual loads and removal efficiencies contain some unknown amount of error. These results indicate that the apparent increase in the wetland's TP removal efficiency from 28 percent during the precontrol period to 45 percent during the postcontrol period can be considered, at least, a minimum (17 percent) improvement in wetland removal of TP.

Orthophosphate

Postcontrol orthophosphate (PO_4) concentrations at the inflow and outflow sites generally were greater than the precontrol concentrations, and outflow concentrations were greater than inflow concentrations during both periods (table 24; fig. 12). Average annual postcontrol inflow and outflow loads were 12 and 7 percent, respectively, larger than precontrol inflow and outflow loads (table 25). The wetland generated and exported PO_4 in quantities greater than those that entered the wetland throughout the study period. The PO_4 outflow loads represented about 12 and 15 percent of total phosphorus outflow loads during the pre- and postcontrol periods, respectively. The exportation of PO_4 from the wetland appeared to decrease slightly; the precontrol removal efficiency of -38 percent increased to a postcontrol value of -31 percent. The average annual PO_4 load generated by the wetland was about 0.6 tons during both periods, however (table 25), and indicated that the concurrent increases in PO_4 loads at the inflow and outflow sites distorted the interpretation of the removal-efficiency values.

Monthly outflow loads exceeded monthly inflow loads, as did the monthly flow-weighted concentrations, throughout the study period (fig. 14). Flow-weighted concentrations at both sites showed an annual pattern of peak concentrations during the summer, and a slight increase during late fall to early winter. The peak summer concentrations reflected an increase in microbial activity and decomposition of organic matter; the

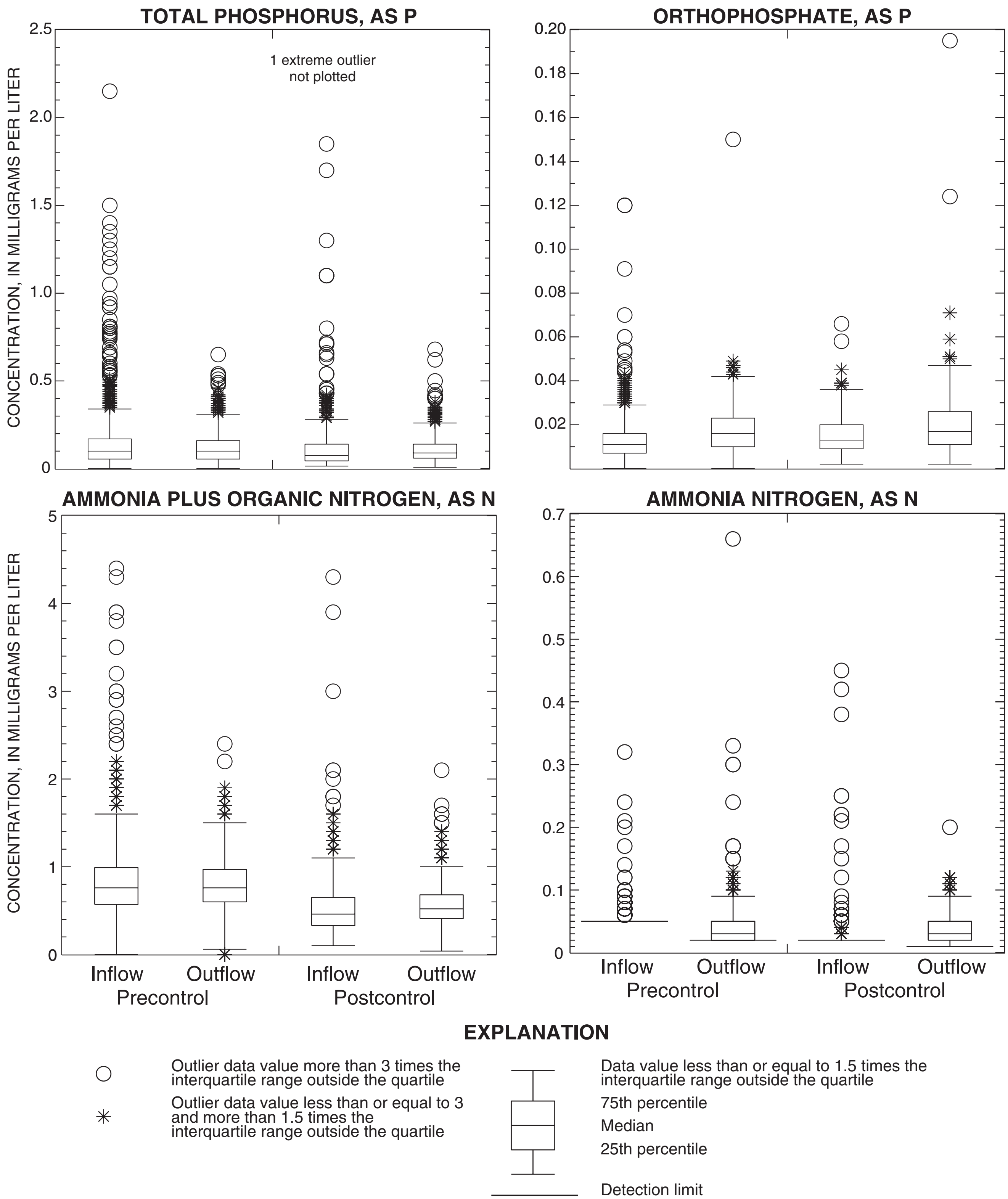


Figure 12. Concentrations of 16 constituents in and discharges associated with surface-water samples collected during pre- and postcontrol periods from inflow (Blossom Road) and outflow (Empire Boulevard) of Ellison Park wetland, Monroe County, N.Y., 1990-2001

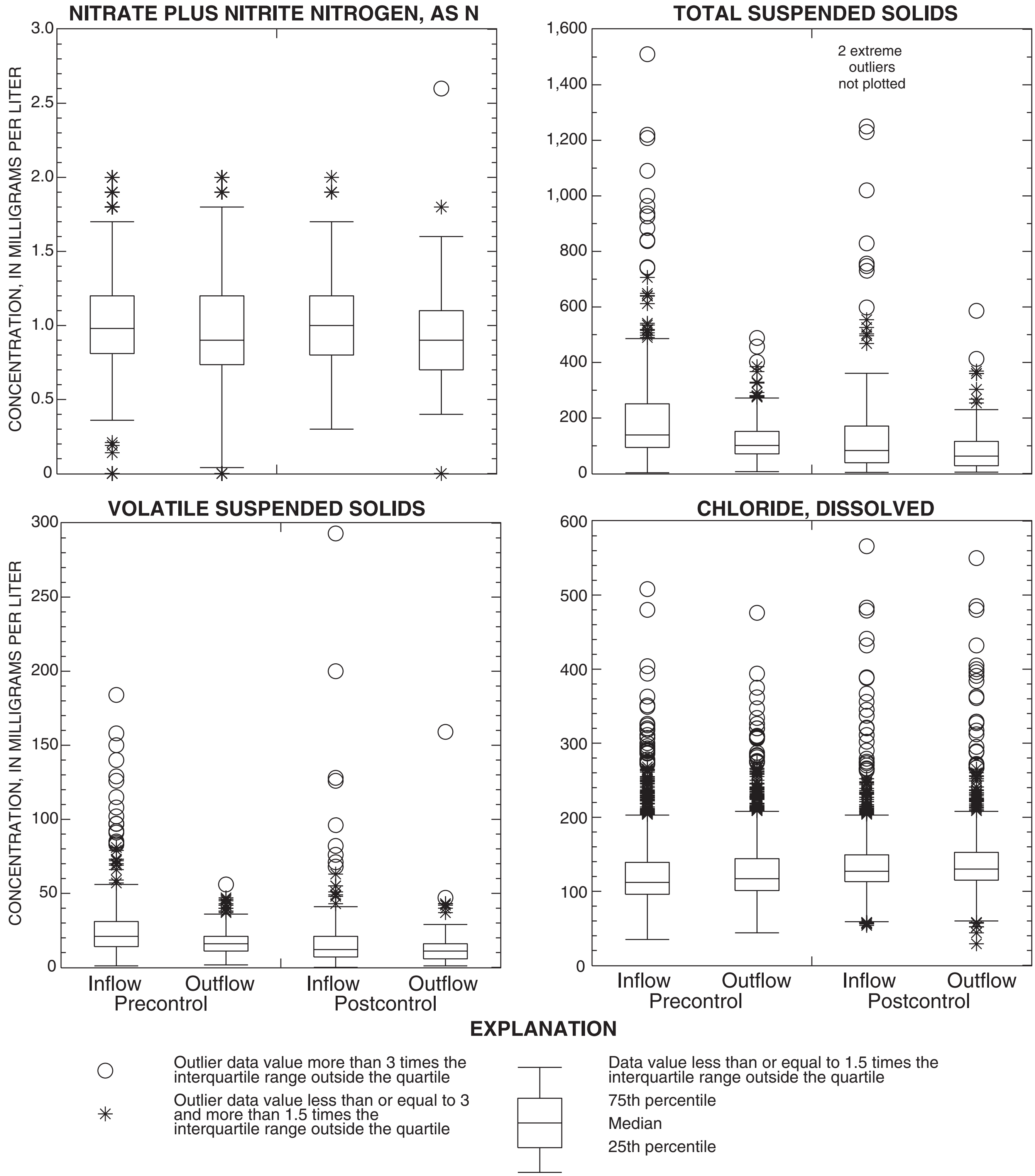


Figure 12 (continued). Concentrations of 16 constituents in and discharges associated with surface-water samples collected during pre- and postcontrol periods from inflow (Blossom Road) and outflow (Empire Boulevard) of Ellison Park wetland, Monroe County, N.Y., 1990-2001

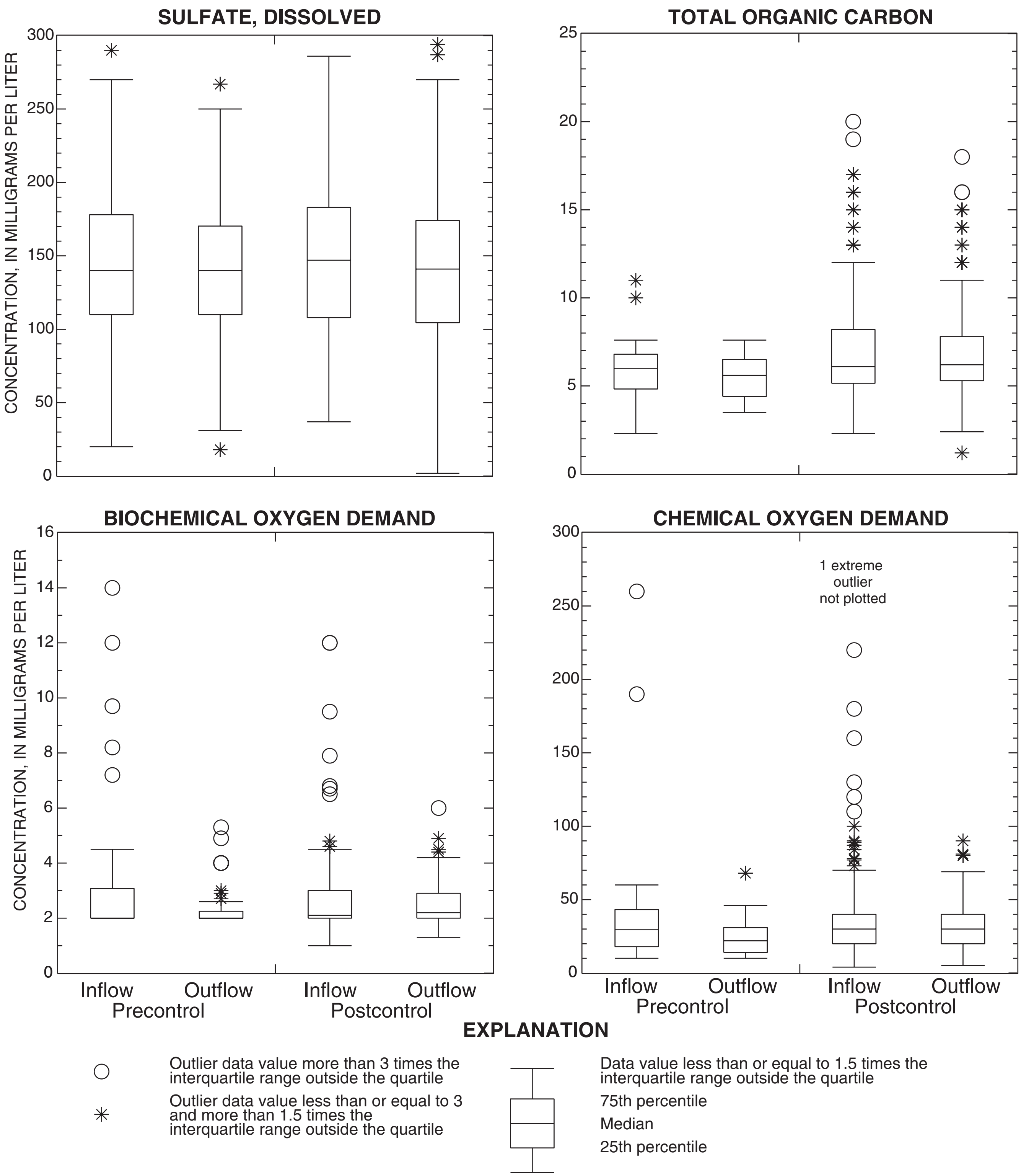


Figure 12 (continued). Concentrations of 16 constituents in and discharges associated with surface-water samples collected during pre- and postcontrol periods from inflow (Blossom Road) and outflow (Empire Boulevard) of Ellison Park wetland, Monroe County, N.Y., 1990-2001

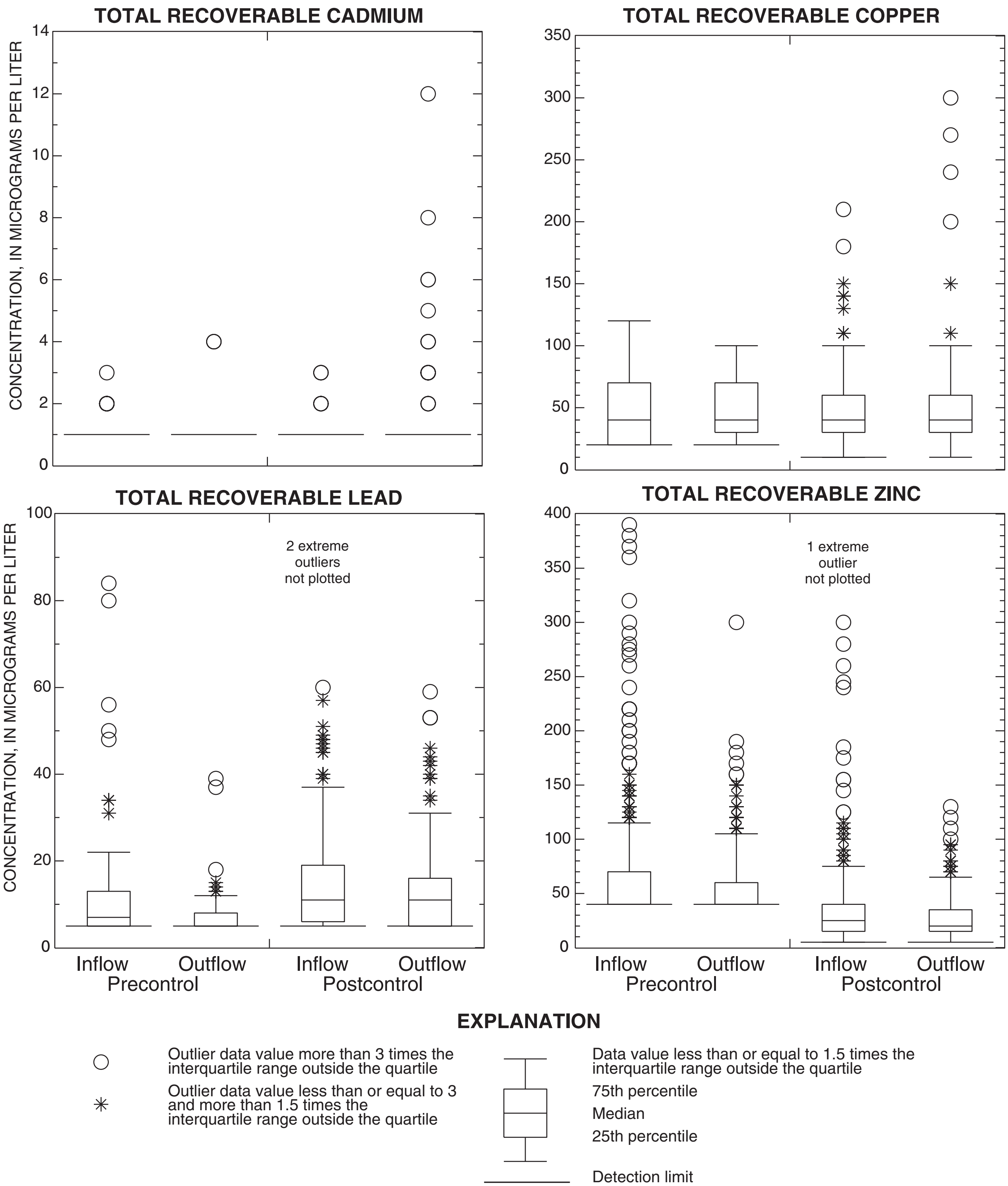
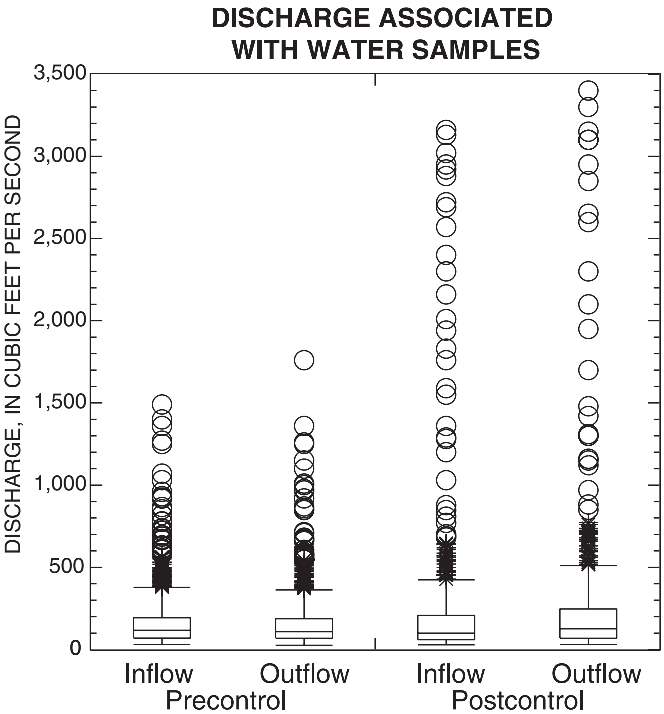


Figure 12 (continued). Concentrations of 16 constituents in and discharges associated with surface-water samples collected during pre- and postcontrol periods from inflow (Blossom Road) and outflow (Empire Boulevard) of Ellison Park wetland, Monroe County, N.Y., 1990-2001



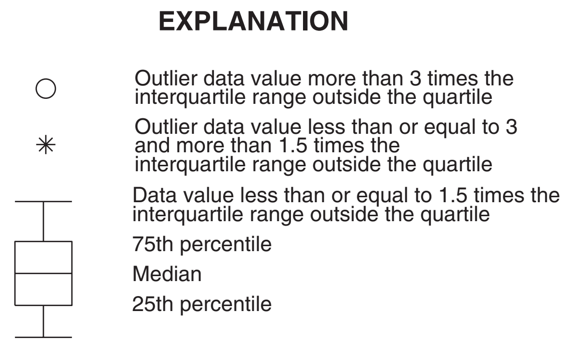


Figure 12 (continued). Concentrations of 16 constituents in and discharges associated with surface-water samples collected during pre- and postcontrol periods from inflow (Blossom Road) and outflow (Empire Boulevard) of Ellison Park wetland, Monroe County, N.Y., 1990-2001

fall-to-winter increase could reflect leaching and decomposition of leaf litter. The large decrease in removal efficiency that coincided with installation of the FCS in February 1997 probably resulted from disturbance and erosion of the channel in the vicinity of the FCS. The percentage of PO_4 exported from the wetland decreased from 1998 through 2001 in a trend similar to that which appeared to be occurring at the end of the precontrol period (late 1994-96; fig. 14). Statistical analyses confirmed that outflow loads were significantly greater than inflow loads, as were the flow-weighted concentrations ($p < 0.0001$; table 26). Adjustment of postcontrol loads and flow-weighted concentrations for the precontrol "wetland effect" revealed no significant difference in PO_4 loads or concentrations that could be attributed to the FCS.

Ammonia-plus-Organic Nitrogen

Postcontrol concentrations of ammonia-plus-organic nitrogen (TKN) generally were less than precontrol concentrations at both sites, and the outflow concentrations exceeded inflow concentrations during the postcontrol period, but were similar to inflow concentrations during the precontrol period, (table 24; fig. 12). Postcontrol average annual loads at the inflow site were 28 percent lower than the precontrol loads, and, at the outflow site, were 24 percent lower (table 25). The wetland's overall removal efficiency decreased from 6.1 percent during the precontrol period to 0.7 percent during the postcontrol period. Postcontrol outflow loads exceeded inflow loads during low-flow periods and also exceeded the loads predicted by the precontrol regression equation, but generally were lower than inflow loads during high-flow months (fig. 15). Monthly removal efficiencies appear to be more negative than positive during the postcontrol period (fig. 15), but the small loads associated with the negative removal efficiencies are offset by the large loads associated with the positive values; hence, a low, but positive, removal efficiency is indicated. Precontrol monthly inflow loads appear similar to monthly outflow loads (fig. 15), and no significant difference is evident ($p = 0.0512$; table 26). Statistically significant differences were noted between postcontrol inflow and outflow loads and flow-weighted concentrations, however ($p = 0.0004$ and 0.0026, respectively). Adjustment for the precontrol "wetland effect" decreased these p values to less than 0.0001 and 0.0023, respectively; this indicates that the FCS caused a significant, although slight, decrease in TKN loads.

Ammonia

Ammonia (NH_3) concentrations at the outflow site exceeded those at the inflow site during both periods. The postcontrol inflow concentrations were lower than the precontrol inflow concentrations, whereas postcontrol outflow concentrations were similar to precontrol outflow concentrations (table 24; fig. 12). Average annual NH_3 loads

Table 25. Average annual inflow and outflow loads of selected constituents, and wetland's removal efficiency for each constituent during pre- and postcontrol periods, Ellison Park wetland, Monroe County, N.Y., 1991-2001

[Inflow site is Blossom Road; outflow site is Empire Boulevard. Loads are in tons. Removal efficiency, in percent, is computed from total (not average annual) loads for the period. Locations are shown in fig. 2.]

Constituent	Precontrol period (water years 1991-96)[1]			Postcontrol period (water years 1997-2001)[1]		
	Inflow load	Outflow load	Removal efficiency	Inflow load	Outflow load	Removal efficiency
Total phosphorus	26.2	19.0	27.6	29.6	16.4	44.6
Orthophosphate	1.69	2.33	- 38.1	1.90	2.50	-31.4
Total nitrogen	275	263	4.3	230	230	-0.1
Ammonia-plus-organic nitrogen	126	118	6.1	90.6	90.0	0.7
Ammonia nitrogen	2.59	4.76	- 84.0	2.07	5.25	-153
Nitrate plus nitrite	149	145	2.7	139	140	-0.6
Total suspended solids	28,500	15,200	46.7	24,500	11,900	51.5
Volatile suspended solids	3,540	2,080	41.0	2,560	1,580	38.1
Chloride	16,800	17,500	- 3.9	18,500	20,200	-8.7
Sulfate	16,000	15,700	2.0	15,100	15,400	-1.8

[1] Water year extends from October 1 of one year through September 30 of the following year.

decreased 20 percent at the inflow site during the postcontrol period, and increased 10 percent at the outflow site (table 25).

Monthly outflow loads were consistently greater than inflow loads throughout the study period, but were substantially lower than predicted by the precontrol regression equation for the postcontrol period (fig. 16). The outflow loads represented 1.8 and 2.3 percent of total nitrogen loads for the pre- and postcontrol periods, respectively. Monthly flow-weighted concentrations in outflow also exceeded those in inflow, and showed a semiannual pattern of peak NH_3 generation during June and July and during December and January of each year. The peak summer concentrations coincided with warm temperatures and high rates of microbial activity, which facilitated NH_3 generation through mineralization of organic matter. Seasonally high water levels also might have contributed to NH_3 loads by promoting diffusion of NH_3 from anaerobic soils. The NH_3 peaks in flow-weighted concentrations occurred at the inflow site—the winter peaks generally were higher than summer peaks—and indicated that NH_3 generation was a basinwide phenomenon and not limited to the wetland. The reason for the midwinter peaks in flow-weighted concentrations is uncertain but could reflect the flushing of NH_3 released from the decomposition of organic matter. Removal efficiencies showed a downward trend (increasing generation and exportation of NH_3) during the precontrol period and an annual pattern of fluctuation within a wide, but fairly stable, range of negative values during the postcontrol period. The statistical analyses of the loads and flow-weighted concentrations confirmed that outflow values were significantly greater than inflow values ($p < 0.0001$), and adjusting the loads and concentrations for the precontrol "wetland effect" did not indicate that the FCS had a significant effect on these variables (table 26).

NH_3 concentrations in the Auto samples showed a negative bias during both periods and at both sites. The mean bias for the precontrol period at the inflow site (-0.005 mg/L) was half as large as that for the postcontrol period (-0.01) and a third as large as the mean bias computed for both periods at the outflow site (-0.015; table 23). The variation in the Auto-to-EWI differences in inflow samples was large for low-flow samples and decreased for high-flow samples; this variation in the outflow samples was fairly constant regardless of flow rate. All loads were underestimated to some unknown degree—the outflow loads more severely than the inflow loads—because Auto concentrations generally were lower than EWI concentrations. Therefore, (1) the actual differences between inflow loads and outflow loads are larger than the calculated differences, especially those during the precontrol period; and (2) the calculated load differences and removal efficiencies can be considered minimum values. This analysis of the biases in NH_3 concentrations did not change the conclusions drawn from the previous analysis of the estimated load data, however—outflow loads would still exceed inflow loads, postcontrol outflow loads would still exceed precontrol outflow loads, and the removal efficiency would still decrease during the postcontrol period (table 25). Also, the NH_3 load would still represent only a small percentage of the total nitrogen load.

Nitrate-plus-Nitrite Nitrogen

Nitrate-plus-nitrite nitrogen (NO_x) concentrations during the postcontrol period were similar to those of the precontrol period at both sites, but the inflow concentrations generally exceeded the outflow concentrations (table 24; fig. 12). Average postcontrol inflow and outflow loads were less than precontrol loads (6.7 and 2.6 percent, respectively), and the removal efficiencies for both periods were close to zero (table 25). Monthly outflow loads were similar to monthly inflow loads throughout the study period, and postcontrol

Table 26. Statistics (p values) related to tests of significant difference between the monthly constituent loads and flow-weighted concentrations, as computed for inflow (Blossom Road) and outflow (Empire Boulevard) of Ellison Park wetland, Monroe County, N.Y., 1990-2001

[Statistics are based on the differences between monthly paired data; that is, inflow (Blossom Road) value minus outflow (Empire Boulevard) value. A p value less than 0.05 suggests a significant difference between inflow and outflow values; the smaller the p value, the stronger the inference of a significant difference. <, less than.]

	Precontrol period (October 1990 through February 1997)			Postcontrol period (March 1997 through September 2001)				
	Wilcoxon Signed Ranks test			Wilcoxon Signed Ranks test				
Constituent	Differences in monthly loads	Differences in monthly flow-weighted concentrations	Greater load (based on monthly flow-weighted concentrations)	Differences in monthly loads	Differences in monthly flow-weighted concentrations	Load differences adjusted by precontrol mean difference	Flow-weighted-concentration differences adjusted by precontrol mean difference	Greater load (based on monthly flow-weighted concentrations)
Total phosphorus	< 0.0001	< 0.0001	inflow	0.0932	0.0078	< 0.0001	0.0012	inflow
Orthophosphate	< .0001	< .0001	outflow	< .0001	< .0001	.4841	.2403	outflow
Total nitrogen	.0017	< .0001	inflow	.0437	< .0001	< .0001	.0489	inflow
Ammonia-plus-organic nitrogen	.0512	.0120	inflow	.0004	.0026	< .0001	.0023	outflow
Ammonia nitrogen	< .0001	< .0001	outflow	< .0001	< .0001	.1180	.0064	outflow
Nitrate plus nitrite	.0178	< .0001	inflow	.7408	< .0001	.1102	.0267	inflow
Total suspended solids	< .0001	< .0001	inflow	< .0001	< .0001	< .0001	.0003	inflow
Volatile suspended solids	< .0001	< .0001	inflow	< .0001	< .0001	< .0001	< .0001	inflow
Chloride	< .0001	< .0001	outflow	< .0001	< .0001	.0014	.3389	outflow
Sulfate	.0004	< .0001	inflow	< .0001	< .0001	< .0001	.1414	inflow

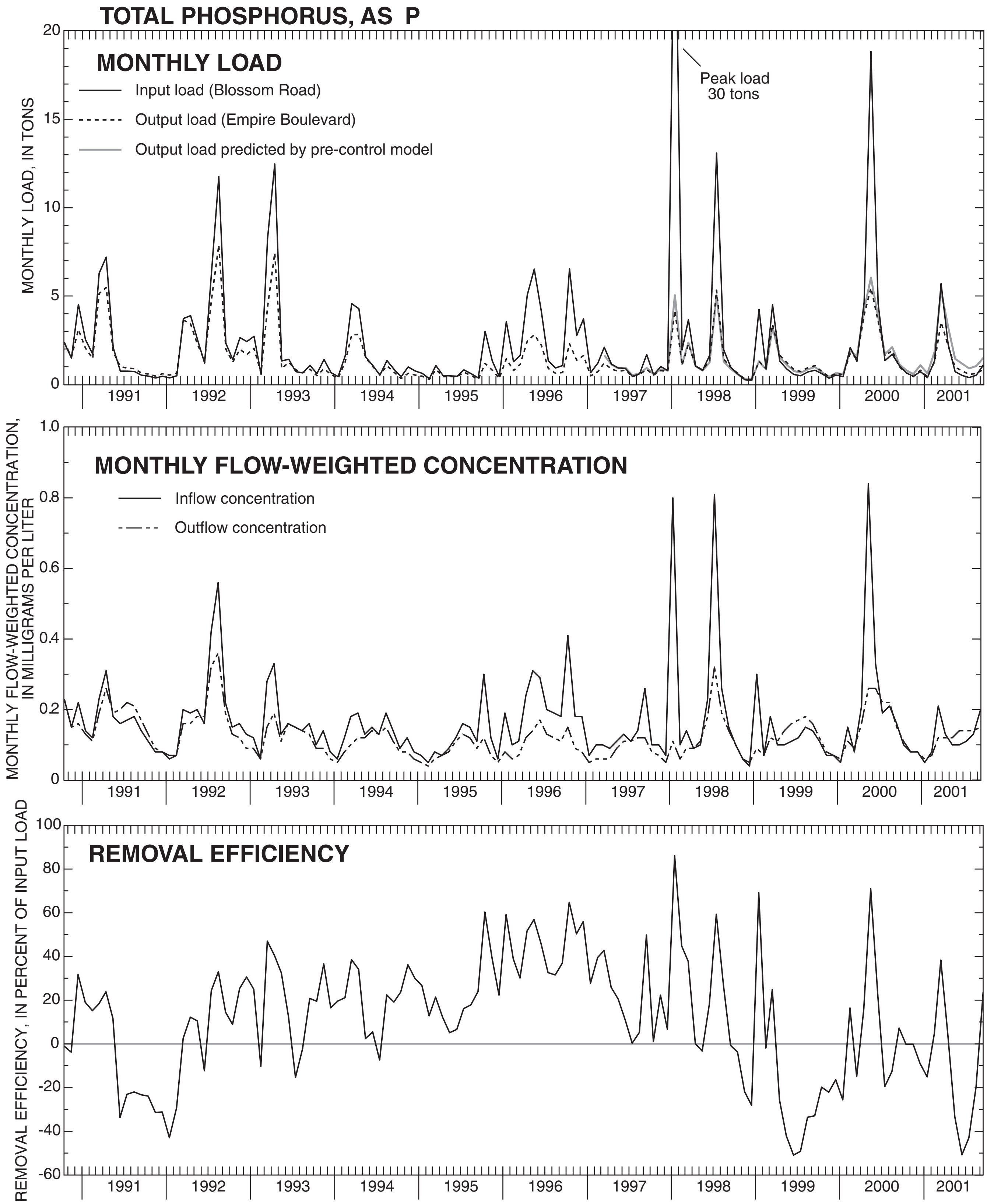


Figure 13. Monthly loads and flow-weighted concentrations, and wetland's removal efficiency for total phosphorus, Ellison Park wetland, Monroe County, N.Y., 1990-2001

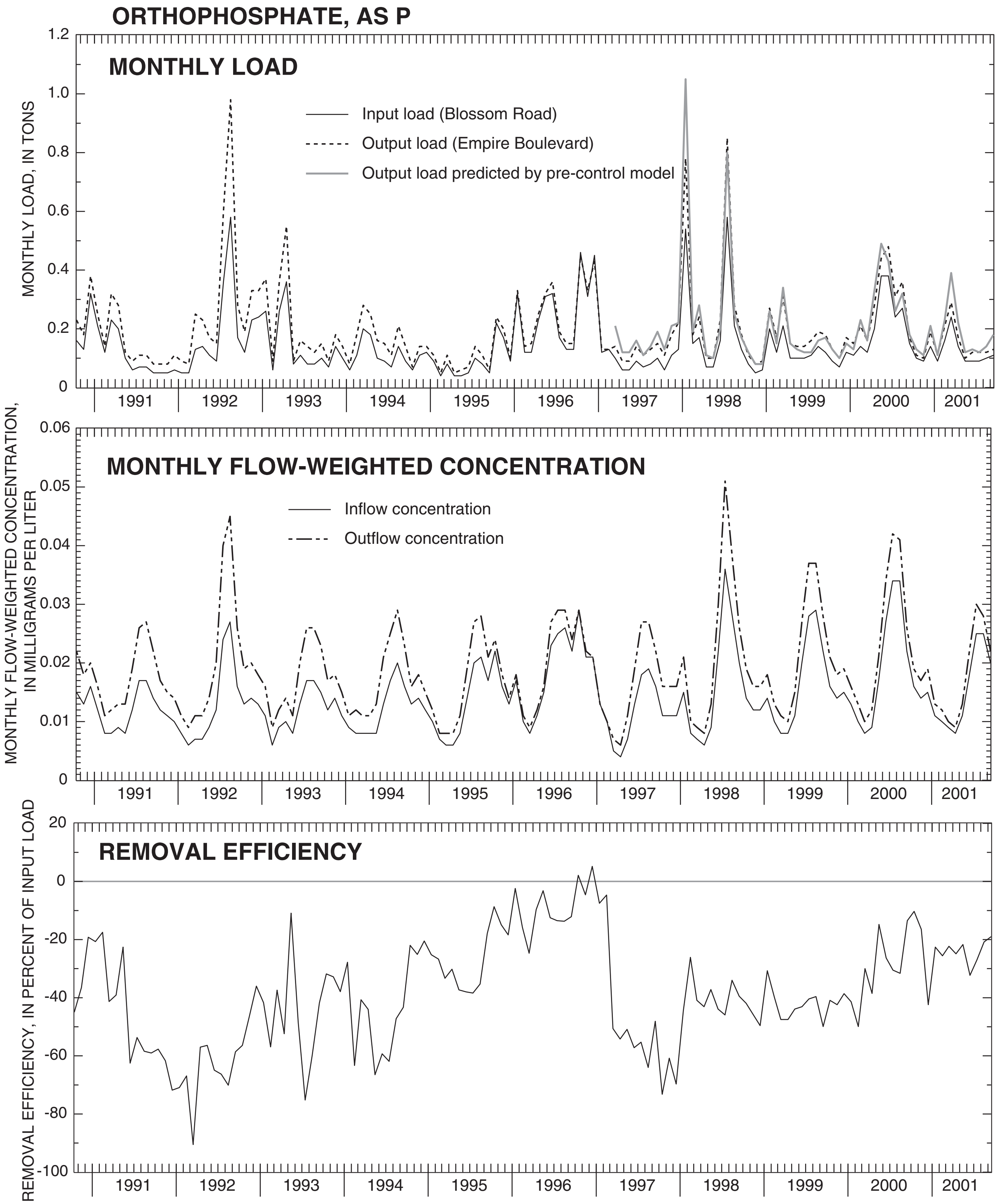


Figure 14. Monthly loads and flow-weighted concentrations, and wetland's removal efficiency for orthophosphate, Ellison Park wetland, Monroe County, N.Y., 1990-2001

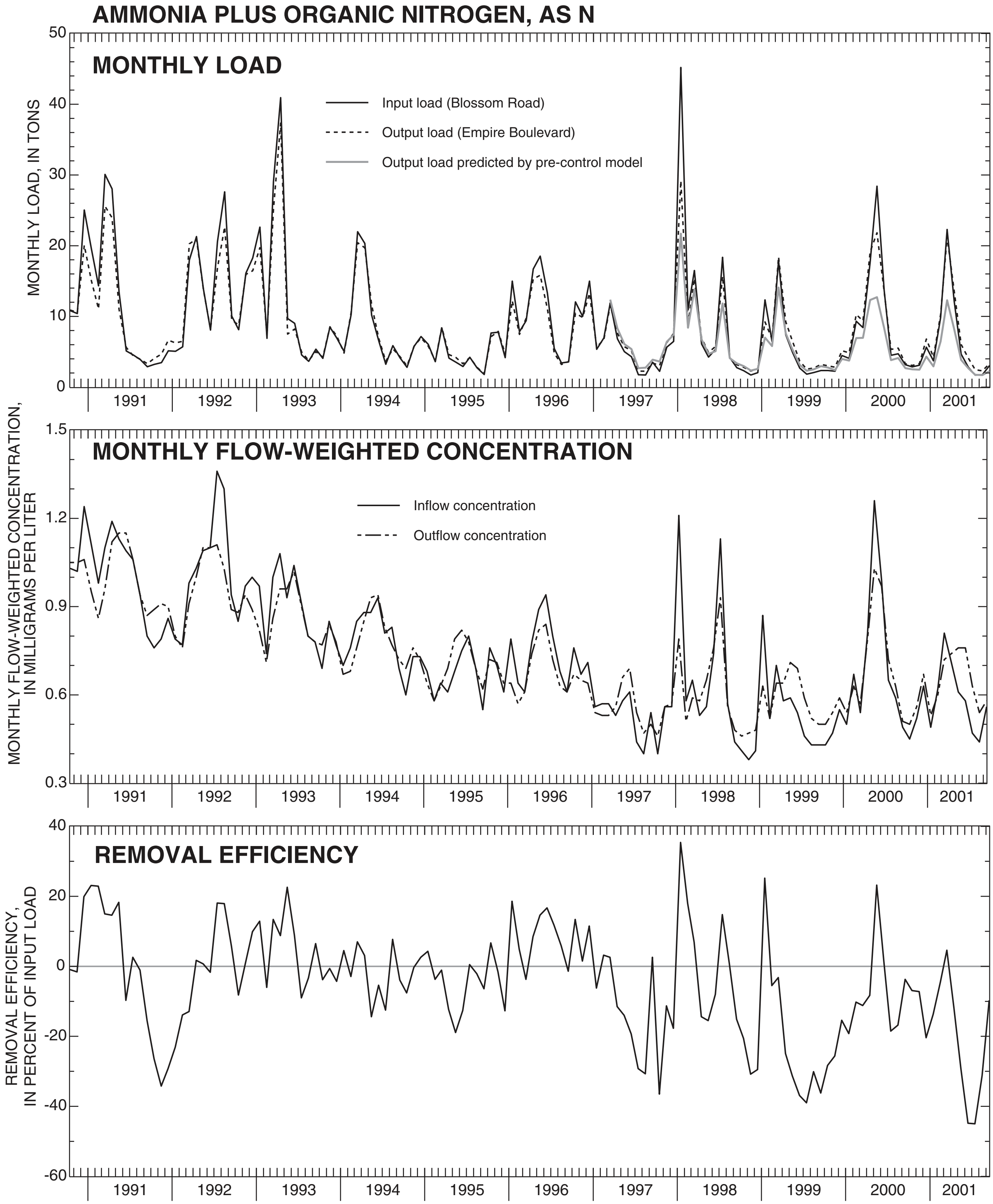


Figure 15. Monthly loads and flow-weighted concentrations, and wetland's removal efficiency for ammonia-plus-organic nitrogen, Ellison Park wetland, Monroe County, N.Y., 1990-2001

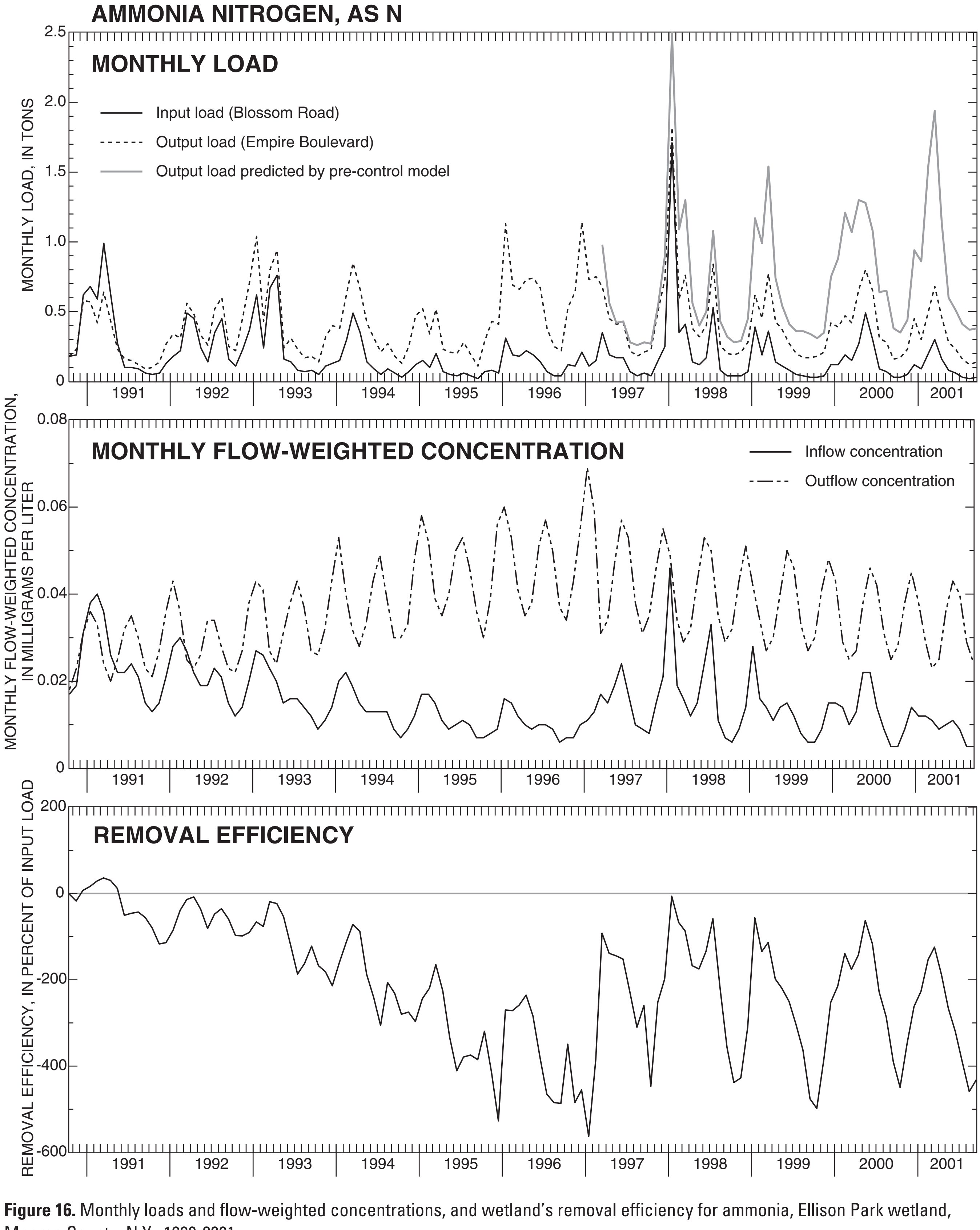


Figure 16. Monthly loads and flow-weighted concentrations, and wetland's removal efficiency for ammonia, Ellison Park wetland, Monroe County, N.Y., 1990-2001

outflow loads were substantially lower than those predicted by the precontrol regression equation (fig. 17). Monthly flow-weighted concentrations at both sites showed an annual pattern of low concentrations during late summer and fall, and high concentrations during the winter. A significant difference ($p < 0.0001$) between monthly flow-weighted concentrations for the two sites was obtained for both periods; but monthly loads for the two sites differed significantly ($p = 0.0178$) only during the precontrol period (table 26). A decrease in maximum flow-weighted concentrations occurred during the postcontrol period at both sites and, thus, could not be attributed to the FCS. Removal-efficiency values reflected an annual pattern as well; with an increase from negative or near-zero in winter to positive values between 10 to 20 percent during the summer. Adjustment of load and flow-weighted-concentration differences by their respective precontrol “wetland effects” indicated that the FCS had little or no effect on NO_x loads (table 26).

A statistically significant bias in NO_x concentrations was noted in outflow samples collected during the postcontrol period only—the concentrations in Auto samples exceeded those measured in EWI samples by 0.034 mg/L on average (table 23). Adjusting the data for this small bias would have slightly decreased the computed postcontrol outflow loads and possibly resulted in a positive removal efficiency during this period, but the change would have been minimal and would not have affected the conclusion that the FCS had little effect on NO_x loads.

Total and Volatile Suspended Solids

Total suspended solids (TSS) outflow concentrations were lower than inflow concentrations during both periods, and postcontrol inflow and outflow concentrations were generally lower than precontrol concentrations (table 24; fig. 12). The average annual outflow load was significantly lower than inflow loads during both periods (table 25). The average annual inflow load decreased 14 percent between the two periods; whereas average annual outflow load decreased 22 percent. Monthly outflow loads were lower than inflow loads during both periods, and the outflow loads during the postcontrol period were smaller than those predicted by the precontrol regression equation (fig. 18). The overall removal efficiency for TSS increased slightly from 47 percent before FCS installation to 52 percent thereafter. The monthly outflow loads and flow-weighted concentrations were significantly less than the inflow loads and flow-weighted concentrations during both periods ($p < 0.0001$; table 26). Adjusting the postcontrol load and flow-weighted-concentration differences by the respective precontrol mean differences yielded mixed results; the significance of the adjusted differences was no greater than those of the unadjusted differences. Therefore, the FCS cannot be said to have contributed to the decrease in TSS loads.

As in the TP analysis, a positive bias was detected in TSS concentrations in the Auto samples from the inflow site during both periods (table 23); therefore, the computed inflow loads and removal efficiencies were erroneously high. The bias varied greatly among low-flow samples—from an 8-percent difference to about a 75-percent difference between Auto and EWI concentrations—and averaged about 54 percent. The bias decreased with increasing discharge, with only an 8-percent difference between Auto (158 mg/L) and EWI (146 mg/L) concentrations for the highest sampled flow of 800 ft^3/s. Therefore, the error in loads presumably was greatest during low flows, when the total load of TSS was low, and least during high flows, when the total load was greatest. The postcontrol bias (9 mg/L) was less than the precontrol bias (33 mg/L). Therefore, the estimated inflow loads exceeded the actual loads by some unknown quantity; the percent error in the postcontrol loads was less than in the precontrol loads; and the actual removal efficiencies were less than those computed from the estimated load data; nevertheless, the indicated 5-percent increase in overall removal efficiency after FCS installation can be considered an improvement in TSS removal.

The decrease in inflow TSS concentrations and loads during the postcontrol period coincided with and might be at least partly attributable to erosion-control projects that were completed on Irondequoit Creek in Linear Park during 1999 and on Allen Creek during 2000 (fig. 1)—both sites are about 3 mi upstream from the wetland. As much as half of the total sediment loading to the Ellison Park wetland has been estimated to have originated from erosion of the sand-and-silt bluffs in Linear Park (Young and Burton, 1993). Stabilization of the streambanks in these reaches had not resulted in a sharp decrease in the total suspended solids loads at the wetland inflow site, presumably because within channel and overbank storage of previously eroded sediment could continue to be sources of sediment loads for several years. Nevertheless, stabilization of these streambanks is expected to result in measurable decreases in future TSS loads.

Volatile suspended solids (VSS) concentrations and loads mimicked TSS patterns during the study period (fig. 19). Postcontrol concentrations and loads were smaller than precontrol values (tables 24 and 25), and outflow concentrations and loads were significantly lower than inflow values ($p < 0.0001$; table 26). Removal efficiency for VSS decreased slightly after FCS installation—from 41 to 38 percent during the pre- and postcontrol periods, respectively (table 25). Adjustment of the loads and flow-weighted concentrations for the precontrol “wetland effect” indicated significant differences between inflow and outflow values ($p < 0.0001$), but none that could be attributed to the FCS. A positive bias in the Auto samples was identified for the inflow site during the precontrol period (table 23). Adjustment of VSS concentrations for this bias would have decreased the inflow load and removal efficiency during the precontrol period by some unknown, but small, amount; thus, the apparent improvement in VSS removal by the wetland would be slightly less than indicated by the estimated loads.

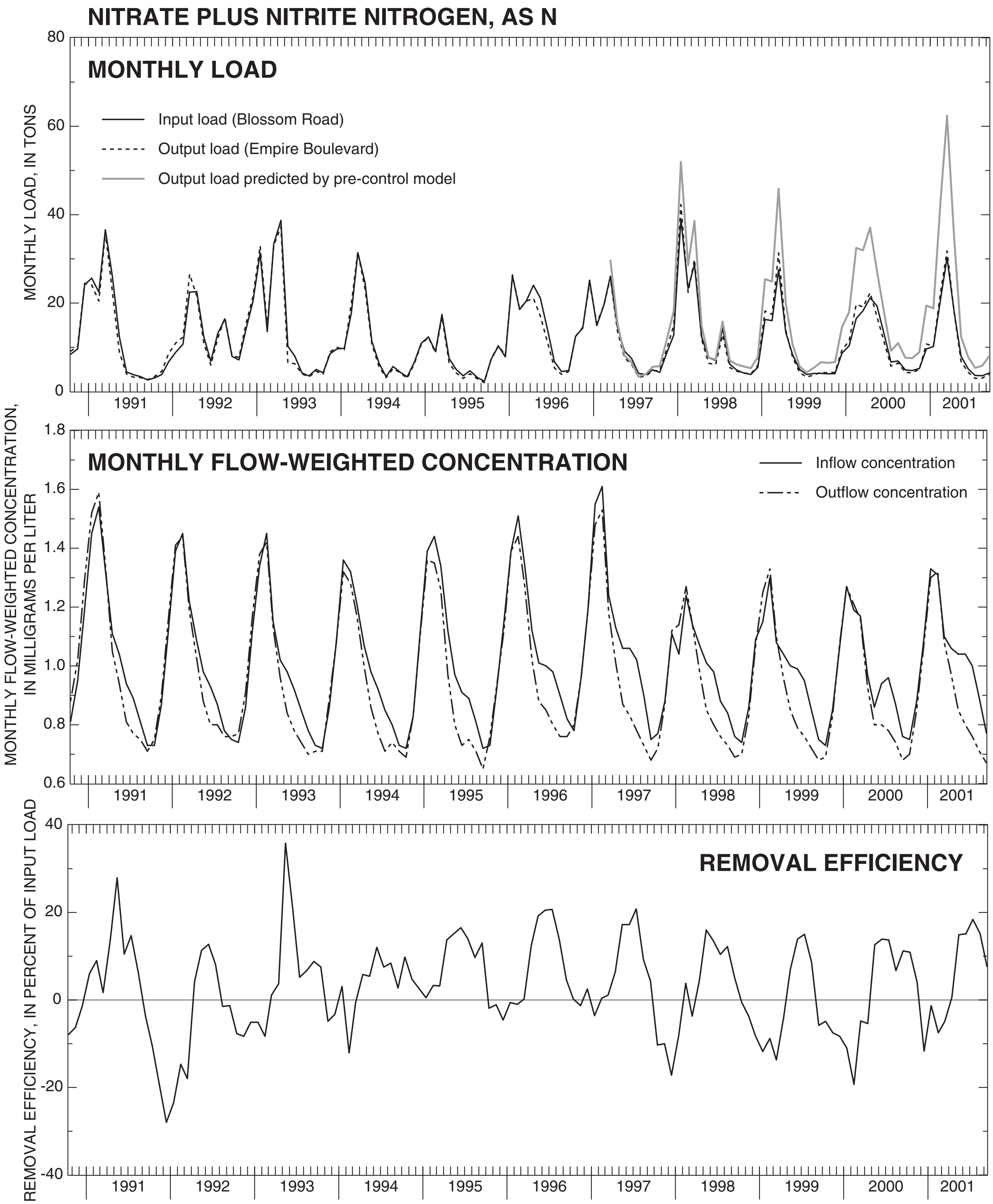


Figure 17. Monthly loads and flow-weighted concentrations, and wetland's removal efficiency for nitrate-plus-nitrite nitrogen, Ellison Park wetland, Monroe County, N.Y., 1990-2001

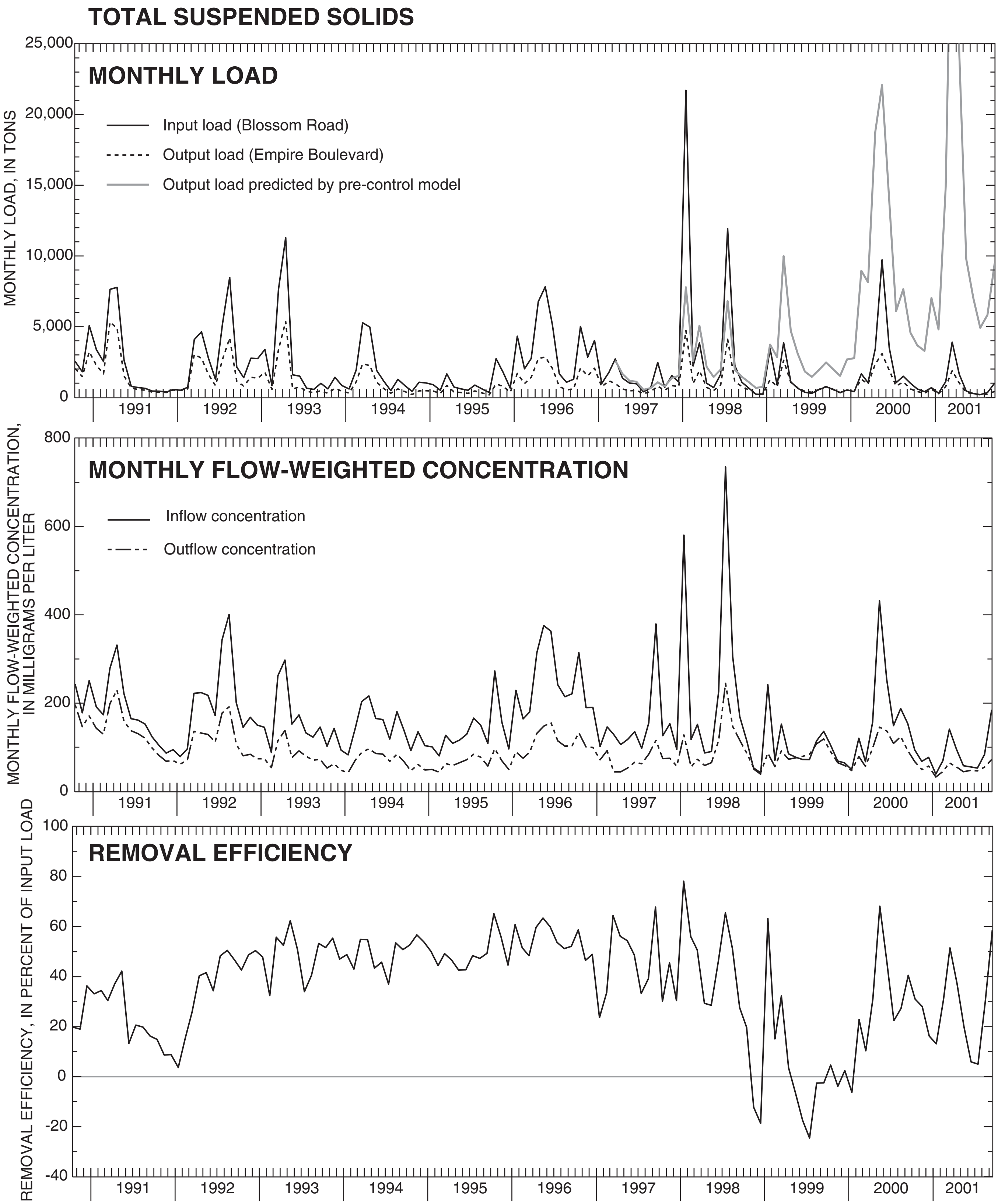


Figure 18. Monthly loads and flow-weighted concentrations, and wetland's removal efficiency for total suspended solids, Ellison Park wetland, Monroe County, N.Y., 1990-2001

Chloride and Sulfate

Chloride (Cl) concentrations at the inflow and outflow sites during a given period were comparable, but the postcontrol concentrations slightly exceeded the precontrol concentrations (table 24; fig. 12). Average annual loads at the inflow and outflow sites during the postcontrol period exceeded the precontrol loads by 10 and 15 percent, respectively (table 25). outflow loads and flow-weighted concentrations were significantly greater than inflow loads ($p < 0.0001$; table 26; fig. 20). The average annual contribution of chloride from the area between the two monitoring sites increased from 3.9 percent during the precontrol period to 8.7 percent during the postcontrol period (table 25). Adjustment of postcontrol flow-weighted-concentration differences by the precontrol mean difference indicated that the FCS did not have a significant effect on chloride loads.

Sulfate concentrations and monthly loads at the inflow were comparable to those at the outflow, and precontrol values were similar to postcontrol values. Monthly inflow flow-weighted concentrations appeared to be greater than the corresponding outflow values, however (table 24; figs. 12 and 21). Average annual loads at the inflow and outflow sites during the precontrol period decreased by 5.6 and 1.9 percent, respectively, during the postcontrol period (table 25). Average annual outflow loads were smaller than inflow loads during the precontrol period, but exceeded them during the postcontrol period, and caused the overall removal efficiency to switch from positive (2.0 percent) to negative (-1.8 percent) after 1997 (table 25). Monthly flow-weighted concentrations showed a more consistent pattern during the study period than did the loads data; the monthly outflow flow-weighted concentrations were slightly, though significantly ($p < 0.0001$; table 26), lower than inflow flow-weighted concentrations during both periods. Adjustment of postcontrol flow-weighted-concentration differences by the precontrol mean difference indicated that the FCS did not have a significant effect on sulfate loads.

Concentrations of Other Constituents

Storm-runoff samples were analyzed for several constituents, including total organic carbon (TOC), biochemical oxygen demand (BOD), chemical oxygen demand (COD), and metals (cadmium, copper, lead, and zinc). Far fewer samples were analyzed for these constituents (except zinc) during the precontrol period than during the postcontrol period (table 24). Analytical procedures were refined during the study period, and lower detection limits were used during the postcontrol period; thus, fewer concentrations of some constituents were censored during the postcontrol period than the precontrol period.

TOC inflow concentrations slightly exceeded outflow concentrations during the precontrol period; median values were 6.0 and 5.6 mg/L, respectively (table 24), but were comparable to the outflow concentrations during the postcontrol period, when median values were 6.1 and 6.2 mg/L, respectively. TOC concentrations were greater at both sites during the postcontrol period than during the precontrol period (table 24; fig. 12).

BOD concentrations were an order of magnitude smaller than COD concentrations, but their concentrations showed similar patterns during the study period. Outflow concentrations generally were lower than inflow concentrations during the precontrol period but were similar to inflow concentrations during the post-control period (table 24; fig. 12). Median BOD concentrations at both sites were below the detection limit during the precontrol period but increased to 2.1 and 2.2 mg/L at the inflow and outflow sites, respectively, during the postcontrol period. Median COD concentrations at the inflow and outflow sites were 30 and 22 mg/L, respectively, during the precontrol period, but were 30 mg/L at both sites during the postcontrol period.

Cadmium concentrations at both sites were below the analytical detection limit during most of the study period. Outflow concentrations, when detected, generally were lower than inflow concentrations.

Copper concentrations in outflow samples were similar to those in inflow samples in both periods (table 24). Median postcontrol concentrations were the same as median precontrol concentrations at both sites—40 mg/L. The 75th-percentile concentrations at both sites were lower during the postcontrol period (60 mg/L) than during the precontrol period (70 mg/L). High outlier values were more frequently detected at both sites during the postcontrol period than during the precontrol period, however (fig. 12).

Lead concentrations in outflow samples generally were lower than in inflow samples during both periods. Median concentrations at the inflow and outflow sites increased from 7 and 5 mg/L, respectively, during the precontrol period to 11 mg/L at both sites during the postcontrol period (table 24). More high outlier values were measured at both sites during the postcontrol period than during the precontrol period (fig. 12).

Zinc concentrations in outflow samples generally were lower than in inflow samples from both periods (fig. 12). Postcontrol concentrations were lower than precontrol concentrations at both sites. Precontrol median concentrations in the inflow samples decreased from 40 to 25 mg/L and, in the outflow samples, from 40 to 20 mg/L (table 24).

The relatively high concentrations of most metals in Irondequoit Creek water samples probably result from urbanization within the basin. Additionally, zinc could be derived from the dissociation of zinc sulfide from dolomite, which underlies part of the basin and is also within the overlying glacial deposits and local soils (Kappel and others, 1986). Depending on the element, decreases in metal concentrations within the Ellison Park wetland could result from several processes, including adsorption on mineral and organic particles, formation of insoluble compounds, sedimentation, ion exchange, bioaccumulation in plant tissues, and biomagnification in aquatic biota (Hem, 1989; Kadlec and Knight, 1996). Concentrations of cadmium, copper, and zinc

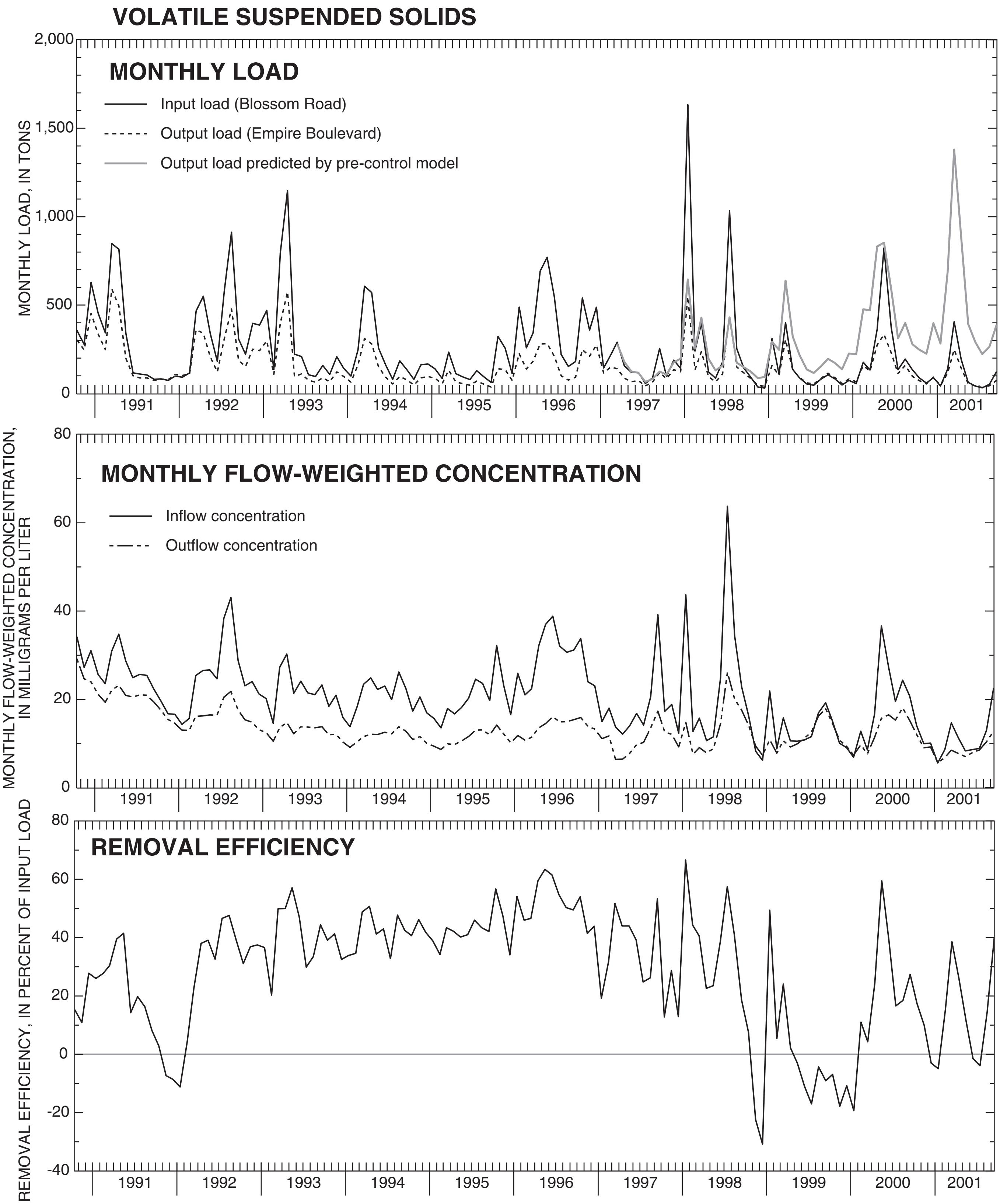


Figure 19. Monthly loads and flow-weighted concentrations, and wetland's removal efficiency for volatile suspended solids, Ellison Park wetland, Monroe County, N.Y., 1990-2001

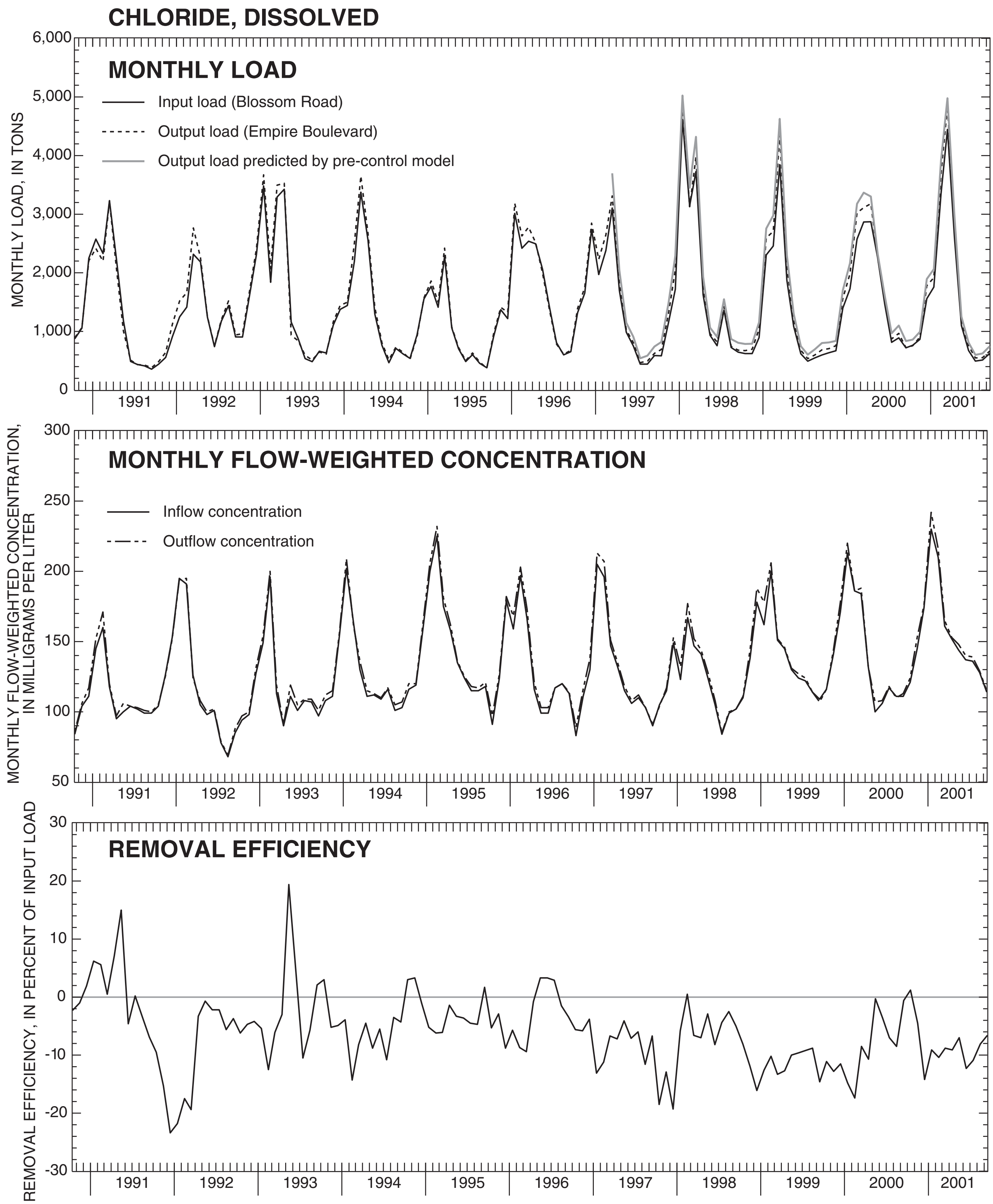


Figure 20. Monthly loads and flow-weighted concentrations, and wetland's removal efficiency for chloride, Ellison Park wetland, Monroe County, N.Y., 1990-2001

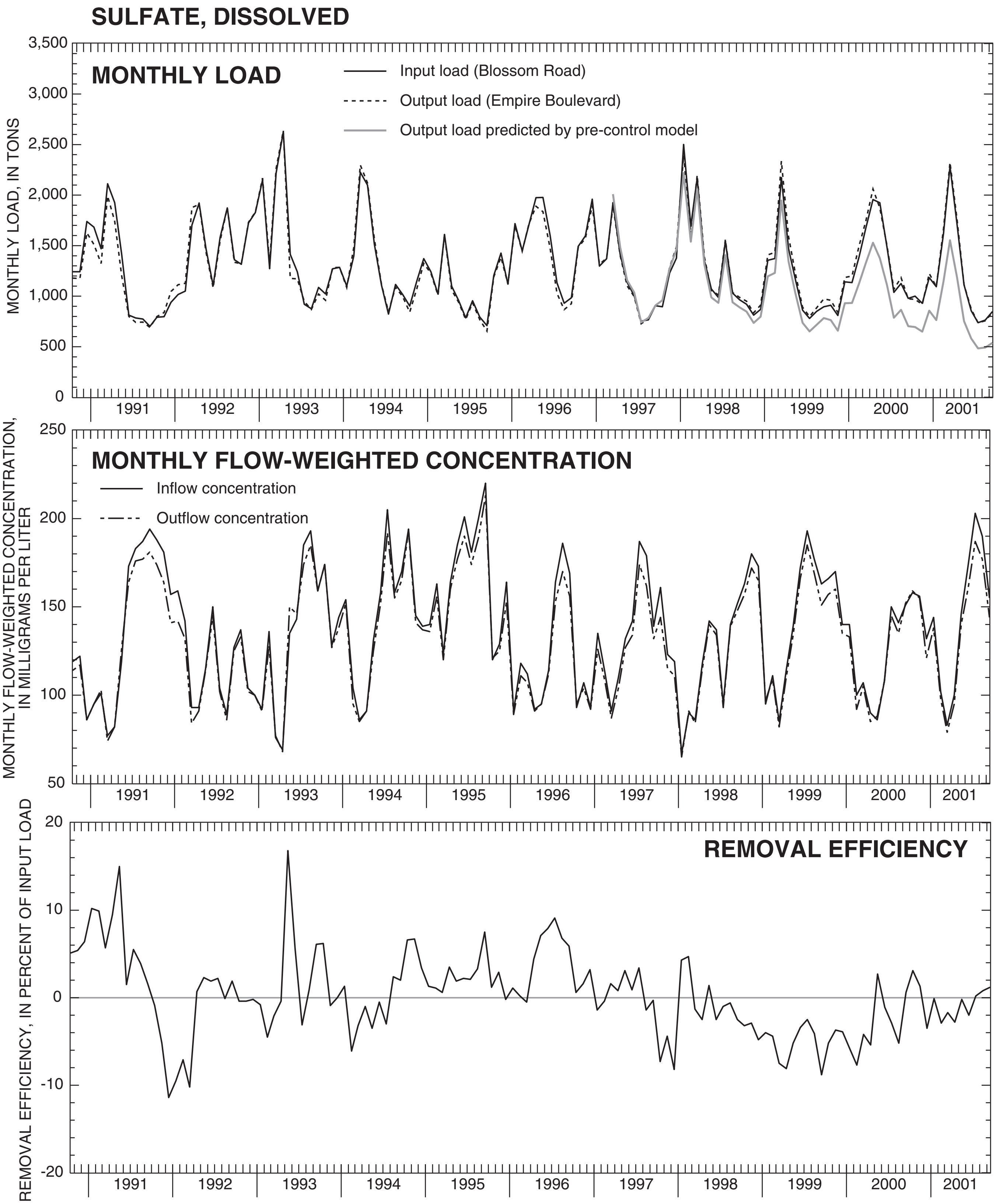


Figure 21. Monthly loads and flow-weighted concentrations, and wetland's removal efficiency for sulfate, Ellison Park wetland, Monroe County, N.Y., 1990-2001

generally were lower than those established by Environment Canada (2003) and New York State Department of Health (2003) as maximum acceptable concentrations in drinking water—5 µg/L for cadmium; 1,000 to 1,300 µg/L for copper; and 5,000 µg/L for zinc—but exceeded the maximum concentrations for the protection of aquatic life—0.017 µg/L for cadmium; 2 to 4 µg/L for copper; and 30 µg/L for zinc—below which negative effects to the most sensitive species of plants and animals would be unexpected (Environment Canada, 2003). Lead concentrations exceeded Canadian and New York State drinking-water (10 to 15 µg/L) and protection-of-aquatic-life guidelines (1 to 7 µg/L) some of the time.

Water-Quality Improvement of Irondequoit Bay

Improvement of the ecological and recreational status of Irondequoit Bay by decreasing the nutrient loads entering the bay has been a management objective of Monroe County since the 1970’s. The MCEHL has been monitoring progress toward this objective by estimating the annual total-phosphorus load entering the bay, and evaluating the bay’s trophic status through two limnological measures of lake fertility—concentrations of phytoplankton chlorophyll and biologically available phosphorus.

Annual Total Phosphorus Load

O’Brien and Gere (1983) estimated the annual load of total phosphorus (TP) entering Irondequoit Bay during 1980-81 to be 15.7 tons, whereas Kappel and others (1986) estimated the annual TP load to be 20.2 tons; this latter value represented a 10-percent decrease from the load estimated at Blossom Road as a result of passage through the wetland. Both analyses estimated that installation of a flow-control structure at the Narrows could achieve a total decrease in TP loads of 24 to 25 percent, or a 15-percent improvement in the wetland’s removal efficiency. The current study found that the average annual TP load entering the bay during the precontrol period was 19.0 tons, which represented a wetland removal efficiency of 28 percent—almost three times the 10-percent effectiveness estimated by Kappel and others (1986). During the postcontrol period, the average annual TP load decreased to 16.4 tons, and the wetland’s removal efficiency increased to 45 percent—a 17-percent improvement in response to the FCS. Although the 1980-81 estimate of the wetland’s removal efficiency (10-percent) was lower than the 1990-96 measured value (28-percent), the measured improvement in removal efficiency (17 percent) that could be attributed to the FCS approximates the improvement estimated by the earlier researchers (15-percent). Therefore, the FCS has met Monroe County’s management objective by this standard.

Trophic Status of the Bay

Chlorophyll_*a* concentrations have been measured in Irondequoit Bay since 1971 (Monroe County Environmental Health Laboratory, written commun., 2002). Water samples are collected biweekly at the deepest point in the bay from May 1 through October 31 at about 3-ft intervals through a 20-ft depth in the epilimnion. Concentrations of chlorophyll_*a* within the water column are averaged for a given day, and the mean of these daily values is used as the summer mean concentration. The biologically available (potential) phosphorus concentration in the bay is computed from the annual total-phosphorus load, the surface-water discharge into the bay, and the bay’s mean depth and surface area (Vollenweider, 1976). A plot of the chlorophyll_*a* and potential phosphorus concentrations in the bay (fig. 22) depicts the bay’s trophic status from 1971 through 2001. Eutrophic conditions prevailed during the 1970’s, when effluent from sewage-treatment plants was discharged into Irondequoit Creek. These conditions improved after the cessation of effluent discharges in 1977 and in response to erosion- and stormwater-control measures implemented by Monroe County since 1980 to decrease chemical loads from nonpoint sources in the basin.

The trophic status of Irondequoit Bay has improved during the last 20 years and has entered the mesotrophic target range of the Irondequoit Bay water-quality management plan (Center for Governmental Research, Inc., 1985) three times during the study period—once before FCS installation (1995) and twice since FCS installation (1999 and 2001) (fig. 22). These apparent annual improvements in trophic status might result from two conditions—lower water levels in the wetland (fig. 4), and fewer and lower-magnitude storms (fig. 9); both would have resulted in lower TP loadings to the bay during these years than during the other study years. The FCS, which had a measurable effect on decreasing TP loads to the bay, presumably still made a contribution toward these improvements during the postcontrol years.

SUGGESTIONS FOR FUTURE MONITORING

The results of this study suggest that future assessments of the effects of the FCS and the other flow modifications could focus on measurements of (1) sedimentation rates, through the annual measurement of sediment thickness at the sedimentation-rate-measurement sites; and (2) the wetland’s removal efficiency of total phosphorus, through continued monitoring of streamflow and water quality at the inflow and outflow sites. These two components of the study documented the most apparent concern (sedimentation upstream from the FCS) and most supportive argument (decreased loads of TP to Irondequoit Bay) for the continued operation of the FCS. Other components of the study contributed little to the assessment of adverse effects of the FCS.

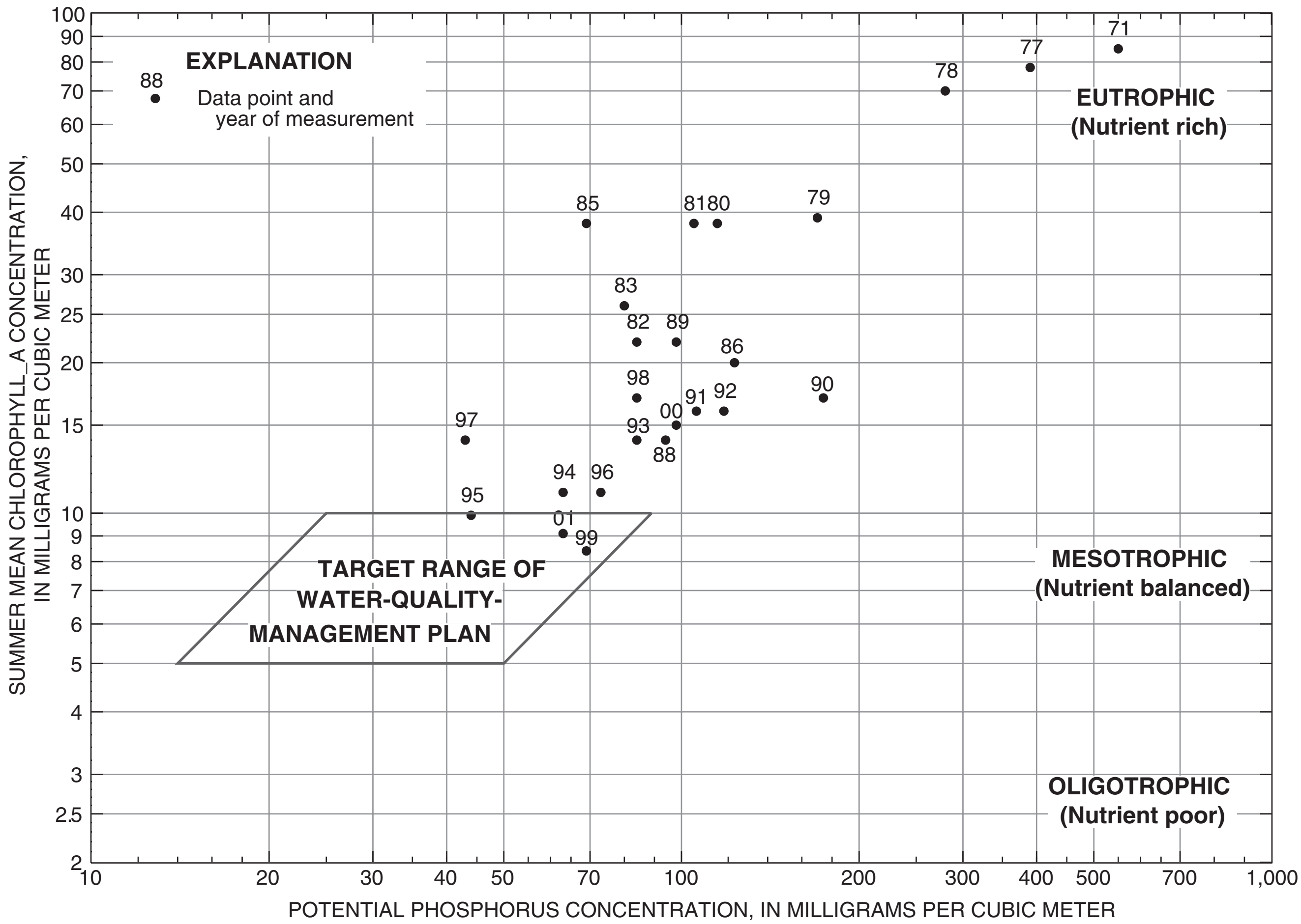


Figure 22. Trophic status of Irondequoit Bay, 1971-2001, as function of measured summer mean concentrations of chlorophyll_*a* and computed concentrations of potential phosphorus

Continued monitoring of the FCS's effects on water levels are probably unnecessary because its flood-causing potential has been adequately evaluated. The ecological monitoring program documented the variable nature of the wetland's biological systems and sediment quality, and the data indicated that identification of any adverse effect of the FCS in the midst of this natural variability would be difficult on the basis of surveys conducted on a greater-than-annual frequency. A monitoring program conducted at a yearly or seasonal frequency might be economically infeasible, however. Therefore, the value of the biological and sediment-quality surveys, which are required by State and Federal regulatory agencies, would be to provide an overview of the general ecological health of the wetland over time, and to detect any egregious change in wetland habitat. A decrease in the areal extent of cattails or open-water areas, or elevated concentrations of chemicals that might be harmful to wetland biota are likely to be detected long before irremediable effects occur, even with a greater-than-annual survey frequency. Once identified, these adverse effects might warrant repetition of surveys at a greater frequency to confirm their existence and extent, and(or) possible mitigation of the effect by removal of the FCS.

SUMMARY AND CONCLUSIONS

Nutrients in Irondequoit Creek, which drains 151 mi^2 of mostly residential and urban land in Monroe County, N.Y., contribute to eutrophic conditions in Irondequoit Bay on Lake Ontario. The Irondequoit Creek basin was studied in the 1979-

81 Nationwide Urban Runoff Program of the U.S. Environmental Protection Agency; one recommendation of that study was to install a flow-control structure (FCS) midway through the Ellison Park wetland, a 423-acre, predominantly cattail (*Typha glauca*) wetland at the mouth of Irondequoit Creek. An 11-year (1990-2001) study was conducted by the U.S. Geological Survey in cooperation with the Monroe County Department of Health to document the effects that flow modifications, which included installation of a FCS in 1997 and increased diversion of stormflows to the backwater areas of the wetland, would have on the wetland's ability to decrease chemical loads transported by Irondequoit Creek into Irondequoit Bay. The FCS was intended to raise the water-surface elevation and thereby increase the dispersal and detention of stormflows in the southern (upstream) half of the wetland and, by increasing sedimentation and time for microbial utilization, decrease the loads of certain constituents, primarily phosphorus, that would otherwise be carried into Irondequoit Bay. Installation of the FCS was permitted by State and Federal wetland-regulatory agencies with the requirement that a comprehensive monitoring program be implemented to assess the effects of the FCS on the wetland's water levels, biota, sedimentation rates, and chemical quality of water and sediment. Periodic surveys of the plant, fish, benthic-macroinvertebrate, and bird communities were conducted. Sedimentation rates were measured, and sediment quality was analyzed. Chemical loads entering and leaving the wetland were estimated and were used to calculate removal-efficiency values before and after FCS installation.

High-flow measurements of the water surface along Irondequoit Creek and discharge at the wetland inflow monitoring site above Blossom Road indicated that as much as 1.5 ft of backwater was possible immediately upstream from the FCS, but that this backwater would not extend as far upstream as Blossom Road (about 13,000 ft). Rather, the backwater observed at Blossom Road was due to constriction of flow by the Browncroft Boulevard bridges, about 5,000 ft upstream from the FCS, or by local debris jams.

The plant surveys, which documented species richness, concentrations of nutrients and metals in cattail tissues, and cattail productivity (above- and below-ground biomass), indicated that observed differences among surveyed years and between the areas upstream and downstream from the FCS were due to seasonal water-level fluctuations that reflected the water-surface elevation of Lake Ontario, rather than water-level control by the FCS. Productivity during a given year apparently was affected by depth of inundation either during the current year, or during the end of the previous year's growing season, when rhizome growth and sprouting of new shoots would be affected by water depth at that time, rather than by the water depth at the beginning of the next growing season. Results indicated no adverse effects to the cattails from either the naturally high water levels that prevail annually during the spring and summer in the wetland, nor from the short-duration elevated water levels that result from FCS operation.

The fish surveys documented the use of the wetland by 44 species, of which 25 to 29 species were found in any given year. Community composition was relatively consistent during the study, but seasonal and year-to-year variations in dominant resident and nonresident species were noted; these variations probably reflected natural or regional population patterns in Lake Ontario and Irondequoit Bay. The FCS did not pose any substantial obstruction to the passage of fish, and did not have other adverse effects on the fish community.

The bird surveys documented the use of the wetland by more than 90 species for breeding, feeding, and migration. The composition of the breeding-bird population during 1996 (precontrol period) differed from that in 2002 (postcontrol period) in that 15 additional breeders were identified in 2002; all breeding species reported in 1996 were also identified in 2002, however. Of wetland ground-nesting species, no evidence of American bittern, American coot, common moorhen, or pied-billed grebe—which had been reported during at least one of the earlier surveys—was found in 2002; least bittern and sora were reported, however. Several other ground-nesting species—ring-necked pheasant, common snipe, blue-winged teal, and American woodcock—were not reported during any of the study-period surveys, but only during the 1980-85 survey, which covered an area much larger than the Ellison Park wetland. Absence or scarcity of specific species was not FCS-related, but was either a regional phenomenon or a result of seasonally high water levels that limited the extent of breeding habitat for some species, and the exposure of mud flats that attracted other species.

The benthic-macroinvertebrate data were of minimal use for evaluating the effect of the FCS because no surveys were conducted after FCS installation; the precontrol results allowed assessment of the ecological quality of the wetland on the basis of biotic indices, however. The benthic-macroinvertebrate taxa in the wetland were typical of low-velocity, organically rich, oxygen-poor aquatic habitats with low taxonomic diversity, and large populations of those species that can survive. Macroinvertebrate-community measures of water quality indicated moderately to severely impaired conditions, whereas species indicators of water quality (such as the high ratio of *Gammarus pseudolimnaeus* to *G. fasciatus*, and low rates of chironomid mouthpart deformities) indicated "clean-water" conditions. Differences between the macroinvertebrate communities in the southern part of the wetland and those in the northern part are attributed to habitat differences, such as substrate composition, water depth, and density of submerged aquatic vegetation.

Sedimentation rates in the area upstream from the FCS increased from 3.0 mm/yr during the precontrol period to 7.3 mm/yr during the postcontrol period, but a similar increase of 2.7 to 4.6 mm/yr was documented in the area downstream from the FCS. The concurrent increase in sediment accumulation downstream from the FCS, and the dynamic pattern of deposition and scour that was documented during the study period indicated that although the FCS undoubtedly played a major role in the apparent postcontrol increase in sedimentation rates upstream from the FCS, other factors, such as the magnitude, frequency, and the timing (season) of peak flows, might also have contributed to this result.

Periodic sediment-quality analyses were used to monitor the concentrations of major and trace elements, polycyclic aromatic hydrocarbons, and organochlorine and organophosphate compounds in sediment samples from three long-term depositional sites in the wetland. The presence of certain constituents, and their concentrations, were fairly typical of a depositional environment in a highly urbanized basin. The concentrations of most constituents showed no substantial fluctuation or consistent upward or downward trend during the years of sampling. The only constituents whose concentrations indicated an upward trend from 1994 through 2001 upstream from the FCS were antimony, calcium, chromium, and possibly mercury and strontium; of these, only chromium showed an upward trend downstream from the FCS. Downward trends were indicated for barium and uranium. Concentrations of many PAHs appeared to decrease in the 1997 and 1999 samples, but the 2001 concentrations were close to those measured in 1994. Conversely, concentrations of the few organochlorine compounds that had detectable concentrations in all four sampling years appeared to increase in the 1997 and 1999 samples, but like many PAHs, had concentrations in 2001 that were comparable to those measured in 1994. Therefore, there was no definable effect of the FCS on sediment quality. Sediment-quality guidelines that are used to assess the ecological quality of substrate environments indicated that the wetland was moderately to severely impaired; this agrees with the benthic-macroinvertebrate biotic indices.

During the precontrol period (1990-96), the wetland was a sink for total phosphorus and total suspended solids (removal efficiencies were 28 and 47 percent, respectively), but had little effect on conservative constituents (chloride and sulfate). The wetland was a source of orthophosphate and ammonia (removal efficiencies were -38 and -84 percent, respectively). The greater loads at the outflow than at the inflow were mainly the result of natural wetland processes, but buried wastes and composting operations in a landfill adjacent to the wetland could also have contributed to these increases.

During the postcontrol period (1997-2001), the wetland continued to be a sink for particulate constituents (removal efficiencies for total phosphorus and total suspended solids were 45 and 52 percent, respectively). The exportation of orthophosphate decreased (the removal efficiency increased to -31 percent), whereas the exportation of ammonia increased (the removal efficiency decreased to -153 percent). The outflow loads of orthophosphate and ammonia represented about 15 and 2.3 percent of total phosphorus and total nitrogen loads, respectively. Changes in the loads of conservative constituents were negligible, and the overall removal efficiencies for other constituents during the precontrol period differed from those of the postcontrol period by no more than 5.4 percent.

Statistical analyses of monthly inflow and outflow loads indicated significant differences between inflow and outflow loads of most constituents during both periods. Adjustment of the load data to remove the effect of hydrologic differences between the precontrol period and the postcontrol period (through calculation of flow-weighted concentrations) indicated similar results. Further adjustment of these data to remove the wetland effect—as defined by the precontrol mean differences between inflow and outflow loads and flow-weighted concentrations—indicated that the FCS contributed significantly to the decrease in total phosphorus loads and slightly to a decrease in ammonia-plus-organic nitrogen loads, but had little or no significant effect on loads of other constituents.

Annual measurements of summer chlorophyll_*a* and calculations of the inflow loads of biologically available phosphorus to monitor the trophic status of Irondequoit Bay indicated that eutrophic conditions improved to near-mesotrophic conditions three times during the study period—once before FCS installation (1995) and twice since FCS installation (1999 and 2001). These apparent annual improvements in trophic status might result in part from lower water levels in the wetland, and fewer and smaller storms during these years than during the other study years; still, the FCS presumably made a contribution toward these improvements during the postcontrol years by retaining phosphorus.

In conclusion, the FCS had no identifiable effect on the wetland biota—plants, fish, or birds—nor on flooding in the vicinity of Blossom Road. Localized increases in sediment accumulation upstream of the FCS were measured, but sediment quality did not appear to be adversely affected by the FCS. The FCS caused a statistically significant decrease in total phosphorus loading to the bay.

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Reports of Biological Studies

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Appendix. Monthly and annual inflow and outflow loads, associated errors, and removal efficiencies for selected constituents, Ellison Park wetland, Monroe County, N.Y., 1996-2001

[Inflow and outflow locations refer to Irondequoit Creek above Blossom Road and at Empire Boulevard, respectively. Load and error values are in tons. Error is standard error of prediction, which, when multiplied by 1.96 and added to and subtracted from the estimated load, provides 95-percent confidence limits of the load estimate. RE, removal efficiency, in percent, is computed from load values prior to rounding to two decimal places. Water year REs are based on total annual loads, not mean monthly values. Positive (negative) RE value indicates net retention in (export from) wetland, respectively. WY, water year. See Coon and others (2000) for 1990-96 data.]

A. Phosphorus Compounds

Month-year	Total phosphorus, as P					Orthophosphate, as P				
	Inflow		Outflow			Inflow		Outflow		
	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE
Oct-96	6.54	2.80	2.30	0.46	64.8	0.46	0.11	0.45	0.06	2.1
Nov-96	2.76	0.84	1.37	0.21	50.3	0.31	0.05	0.33	0.03	-4.6
Dec-96	3.71	1.10	1.63	0.24	56.0	0.45	0.07	0.43	0.04	5.2
Jan-97	0.67	0.12	0.48	0.05	27.7	0.12	0.01	0.13	0.01	-7.5
Feb-97	1.21	0.31	0.73	0.10	39.5	0.13	0.02	0.13	0.01	-4.7
Mar-97	2.10	0.33	1.21	0.13	42.7	0.10	0.01	0.14	0.01	-50.6
Apr-97	1.15	0.19	0.86	0.10	25.8	0.06	0.00	0.09	0.01	-54.2
May-97	0.93	0.19	0.74	0.09	20.5	0.06	0.01	0.09	0.01	-50.9
Jun-97	0.90	0.15	0.80	0.09	10.9	0.09	0.01	0.14	0.01	-57.2
Jul-97	0.45	0.07	0.45	0.05	0.3	0.07	0.01	0.11	0.01	-55.3
Aug-97	0.60	0.13	0.57	0.08	5.2	0.08	0.01	0.13	0.01	-64.0
Sep-97	1.69	0.67	0.85	0.15	49.8	0.10	0.01	0.15	0.02	-48.1
WY97 total	**22.71**	**6.91**	**11.99**	**1.74**	**47.2**	**2.03**	**0.32**	**2.33**	**0.23**	**-14.7**
Oct-97	0.54	0.10	0.53	0.06	1.0	0.06	0.01	0.11	0.01	-73.2
Nov-97	1.01	0.18	0.78	0.09	22.3	0.11	0.01	0.18	0.02	-60.8
Dec-97	0.78	0.11	0.73	0.07	6.6	0.13	0.01	0.22	0.02	-69.7
Jan-98	30.02	13.35	4.18	1.03	86.1	0.54	0.08	0.78	0.11	-44.4
Feb-98	1.95	0.39	1.08	0.13	44.8	0.15	0.01	0.18	0.02	-26.1
Mar-98	3.66	0.61	2.28	0.24	37.8	0.17	0.01	0.24	0.02	-40.9
Apr-98	1.04	0.16	1.04	0.11	0.1	0.07	0.01	0.10	0.01	-43.1
May-98	0.79	0.15	0.82	0.10	-3.3	0.07	0.01	0.10	0.01	-37.2
Jun-98	1.64	0.47	1.34	0.21	18.2	0.16	0.02	0.23	0.02	-43.9
Jul-98	13.09	5.31	5.33	1.13	59.3	0.58	0.07	0.85	0.11	-45.9
Aug-98	1.95	0.60	1.42	0.21	27.1	0.21	0.02	0.28	0.03	-34.0
Sep-98	0.96	0.16	0.96	0.11	-0.7	0.13	0.01	0.18	0.01	-39.5
WY98 total	**57.44**	**21.60**	**20.49**	**3.49**	**64.3**	**2.38**	**0.27**	**3.45**	**0.37**	**-44.8**
Oct-98	0.59	0.11	0.62	0.07	-3.8	0.08	0.01	0.11	0.01	-41.9
Nov-98	0.26	0.03	0.31	0.03	-21.9	0.05	0.00	0.08	0.01	-45.8
Dec-98	0.22	0.03	0.28	0.03	-28.1	0.06	0.00	0.09	0.01	-49.6
Jan-99	4.24	1.98	1.30	0.27	69.2	0.20	0.03	0.27	0.03	-30.7
Feb-99	0.87	0.14	0.89	0.10	-1.9	0.12	0.01	0.17	0.01	-39.8
Mar-99	4.51	0.90	3.39	0.41	24.9	0.21	0.02	0.31	0.03	-47.5
Apr-99	1.32	0.20	1.66	0.17	-25.6	0.10	0.01	0.15	0.01	-47.5
May-99	0.88	0.13	1.24	0.13	-42.1	0.10	0.01	0.14	0.01	-43.9
Jun-99	0.56	0.08	0.84	0.09	-50.9	0.10	0.01	0.14	0.01	-43.1
Jul-99	0.49	0.06	0.73	0.07	-49.2	0.11	0.01	0.16	0.01	-40.4
Aug-99	0.70	0.11	0.94	0.10	-33.6	0.14	0.01	0.19	0.01	-39.6
Sep-99	0.80	0.14	1.06	0.13	-32.9	0.12	0.01	0.18	0.02	-49.9
WY99 total	**15.44**	**3.93**	**13.26**	**1.61**	**14.1**	**1.40**	**0.13**	**1.99**	**0.17**	**-42.4**

Appendix. Monthly and annual inflow and outflow loads, associated errors, and removal efficiencies for selected constituents, Ellison Park wetland, Monroe County, N.Y., 1996-2001 (continued)

A. Phosphorus Compounds (continued)

Month-year	Total phosphorus, as P					Orthophosphate, as P				
	Inflow		Outflow			Inflow		Outflow		
	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE
Oct-99	0.60	0.12	0.72	0.09	-19.8	0.09	0.01	0.13	0.01	-40.9
Nov-99	0.35	0.07	0.43	0.06	-22.1	0.07	0.01	0.10	0.01	-42.4
Dec-99	0.53	0.10	0.62	0.08	-16.4	0.12	0.01	0.17	0.01	-38.6
Jan-00	0.44	0.08	0.55	0.07	-25.6	0.11	0.01	0.15	0.01	-41.4
Feb-00	2.09	0.56	1.75	0.30	16.4	0.14	0.02	0.21	0.02	-49.9
Mar-00	1.30	0.18	1.49	0.15	-15.0	0.12	0.01	0.16	0.01	-30.0
Apr-00	4.65	0.97	3.92	0.49	15.7	0.20	0.02	0.28	0.02	-38.5
May-00	18.84	8.95	5.47	1.05	71.0	0.38	0.05	0.44	0.05	-14.8
Jun-00	4.55	1.22	3.57	0.55	21.6	0.38	0.04	0.48	0.05	-26.3
Jul-00	1.34	0.28	1.60	0.20	-19.6	0.24	0.02	0.31	0.03	-30.5
Aug-00	1.71	0.34	1.92	0.23	-12.7	0.27	0.02	0.36	0.03	-31.6
Sep-00	1.06	0.23	0.99	0.11	7.2	0.15	0.01	0.17	0.01	-13.5
WY00 total	**37.46**	**13.10**	**23.02**	**3.38**	**38.5**	**2.27**	**0.22**	**2.95**	**0.27**	**-29.8**
Oct-00	0.66	0.11	0.66	0.07	-0.2	0.10	0.01	0.11	0.01	-10.3
Nov-00	0.45	0.08	0.45	0.05	-0.2	0.09	0.01	0.10	0.01	-16.5
Dec-00	0.73	0.19	0.80	0.14	-9.0	0.14	0.01	0.19	0.02	-42.4
Jan-01	0.38	0.08	0.44	0.06	-15.1	0.09	0.01	0.11	0.01	-22.6
Feb-01	1.34	0.24	1.28	0.15	4.8	0.16	0.01	0.21	0.02	-25.6
Mar-01	5.70	1.27	3.52	0.45	38.3	0.24	0.02	0.29	0.02	-22.3
Apr-01	2.27	0.37	2.18	0.25	3.9	0.14	0.01	0.17	0.01	-24.9
May-01	0.72	0.11	0.97	0.10	-33.5	0.09	0.01	0.10	0.01	-21.7
Jun-01	0.50	0.07	0.76	0.08	-50.8	0.09	0.01	0.12	0.01	-32.3
Jul-01	0.39	0.06	0.56	0.06	-43.0	0.09	0.01	0.12	0.01	-27.0
Aug-01	0.52	0.09	0.62	0.07	-19.7	0.10	0.01	0.12	0.01	-20.8
Sep-01	1.11	0.41	0.85	0.15	23.6	0.11	0.01	0.13	0.02	-19.0
WY01 total	**14.78**	**3.08**	**13.07**	**1.64**	**11.5**	**1.42**	**0.13**	**1.77**	**0.16**	**-24.2**
Period of record WY97-01	**147.83**	**48.61**	**81.84**	**11.86**	**44.6**	**9.50**	**1.06**	**12.48**	**1.20**	**-31.4**

Appendix. Monthly and annual inflow and outflow loads, associated errors, and removal efficiencies for selected constituents, Ellison Park wetland, Monroe County, N.Y., 1996-2001

[Inflow and outflow locations refer to Irondequoit Creek above Blossom Road and at Empire Boulevard, respectively. Load and error values are in tons. Error is standard error of prediction, which, when multiplied by 1.96 and added to and subtracted from the estimated load, provides 95-percent confidence limits of the load estimate. RE, removal efficiency, in percent, is computed from load values prior to rounding to two decimal places. Water year REs are based on total annual loads, not mean monthly values. Positive (negative) RE value indicates net retention in (export from) wetland, respectively. WY, water year. See Coon and others (2000) for 1990-96 data.]

B. Nitrogen Compounds

Month-year	Total nitrogen			Nitrate plus nitrite					Ammonia-plus-organic nitrogen					Ammonia nitrogen				
	Inflow	Outflow		Inflow		Outflow			Inflow		Outflow			Inflow		Outflow		
	Load	Load	RE	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE
Oct-96	24.55	22.92	6.7	12.49	0.88	12.46	0.72	0.2	12.06	1.91	10.45	1.09	13.4	0.12	0.05	0.53	0.12	-350
Nov-96	24.36	24.40	-0.2	14.36	0.77	14.55	0.68	-1.3	10.00	1.14	9.85	0.81	1.5	0.11	0.03	0.64	0.12	-485
Dec-96	40.17	37.81	5.9	25.18	1.29	24.54	1.09	2.5	14.99	1.63	13.27	1.06	11.5	0.21	0.05	1.14	0.20	-455
Jan-97	20.27	21.13	-4.3	14.91	0.72	15.45	0.65	-3.6	5.36	0.48	5.69	0.40	-6.2	0.11	0.02	0.73	0.12	-563
Feb-97	26.56	26.26	1.1	19.61	1.09	19.53	0.93	0.4	6.95	0.76	6.73	0.54	3.2	0.15	0.04	0.75	0.14	-384
Mar-97	37.98	37.37	1.6	26.07	1.02	25.77	0.94	1.1	11.91	1.21	11.60	0.98	2.6	0.35	0.09	0.68	0.08	-92
Apr-97	21.38	21.24	0.7	14.54	0.59	13.61	0.52	6.4	6.84	0.74	7.63	0.68	-11.5	0.19	0.05	0.45	0.06	-139
May-97	14.18	13.31	6.1	9.16	0.37	7.59	0.29	17.2	5.02	0.61	5.72	0.56	-14.0	0.17	0.05	0.41	0.05	-145
Jun-97	11.81	11.35	3.9	7.50	0.30	6.21	0.24	17.2	4.31	0.50	5.14	0.49	-19.3	0.17	0.05	0.42	0.05	-152
Jul-97	5.87	5.52	5.9	4.11	0.16	3.25	0.12	20.8	1.76	0.19	2.27	0.21	-29.2	0.07	0.02	0.22	0.03	-228
Aug-97	5.57	5.75	-3.2	3.84	0.16	3.49	0.14	9.2	1.73	0.24	2.26	0.24	-30.7	0.04	0.01	0.18	0.02	-310
Sep-97	8.43	8.13	3.6	4.91	0.25	4.70	0.22	4.3	3.52	0.77	3.43	0.48	2.6	0.06	0.03	0.21	0.04	-260
WY97 total	**241.13**	**235.20**	**2.5**	**156.68**	**7.60**	**151.16**	**6.53**	**3.5**	**84.45**	**10.17**	**84.05**	**7.53**	**0.5**	**1.75**	**0.49**	**6.37**	**1.04**	**-265**
Oct-97	6.56	7.82	-19.2	4.32	0.17	4.77	0.18	-10.3	2.24	0.27	3.05	0.29	-36.5	0.04	0.01	0.23	0.03	-447
Nov-97	14.47	15.99	-10.5	8.82	0.36	9.70	0.37	-10.0	5.65	0.65	6.29	0.60	-11.3	0.15	0.04	0.51	0.07	-252
Dec-97	19.24	22.58	-17.3	12.77	0.48	14.97	0.53	-17.2	6.47	0.65	7.61	0.64	-17.7	0.25	0.06	0.73	0.08	-198
Jan-98	84.25	71.45	15.2	39.06	2.08	42.28	2.22	-8.2	45.19	9.54	29.18	4.68	35.4	1.71	0.82	1.82	0.34	-6
Feb-98	34.22	31.37	8.3	23.30	0.93	22.42	0.84	3.8	10.92	1.30	8.95	0.87	18.0	0.35	0.10	0.59	0.08	-67
Mar-98	45.26	45.19	0.2	28.77	1.07	29.84	1.03	-3.7	16.49	1.74	15.36	1.33	6.9	0.41	0.10	0.76	0.09	-87
Apr-98	18.45	18.77	-1.8	12.35	0.46	11.80	0.41	4.5	6.10	0.63	6.97	0.59	-14.4	0.14	0.04	0.38	0.05	-168
May-98	11.89	11.32	4.8	7.64	0.29	6.41	0.24	16.0	4.25	0.51	4.91	0.47	-15.5	0.12	0.03	0.32	0.04	-175
Jun-98	12.39	11.87	4.2	7.04	0.32	6.09	0.26	13.5	5.35	0.88	5.78	0.68	-8.0	0.17	0.07	0.40	0.06	-134
Jul-98	32.65	28.45	12.9	14.29	0.72	12.80	0.63	10.4	18.36	3.63	15.65	2.39	14.8	0.53	0.24	0.84	0.15	-59
Aug-98	10.38	9.59	7.6	6.18	0.27	5.43	0.22	12.2	4.19	0.68	4.16	0.46	0.7	0.08	0.03	0.26	0.04	-216
Sep-98	7.60	7.77	-2.2	4.84	0.19	4.59	0.17	5.1	2.76	0.31	3.18	0.29	-15.1	0.04	0.01	0.20	0.02	-356
WY98 total	**297.36**	**282.18**	**5.1**	**169.39**	**7.34**	**171.08**	**7.08**	**-1.0**	**127.97**	**20.80**	**111.10**	**13.28**	**13.2**	**3.99**	**1.55**	**7.05**	**1.04**	**-77**

Appendix. Monthly and annual inflow and outflow loads, associated errors, and removal efficiencies for selected constituents, Ellison Park wetland, Monroe County, N.Y., 1996-2001 (continued)

B. Nitrogen Compounds (continued)

Month-year	Total nitrogen			Nitrate plus nitrite					Ammonia-plus-organic nitrogen					Ammonia nitrogen				
	Inflow	Outflow		Inflow		Outflow			Inflow		Outflow			Inflow		Outflow		
	Load	Load	RE	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE
Oct-98	6.51	7.01	-7.7	4.19	0.16	4.21	0.15	-0.5	2.32	0.27	2.80	0.25	-20.6	0.04	0.01	0.19	0.02	-438
Nov-98	5.61	6.28	-12.1	3.87	0.14	4.02	0.14	-3.7	1.73	0.17	2.27	0.19	-30.8	0.04	0.01	0.21	0.03	-427
Dec-98	7.53	8.59	-14.1	5.47	0.21	5.93	0.21	-8.3	2.06	0.22	2.66	0.23	-29.5	0.07	0.02	0.28	0.03	-310
Jan-99	28.64	27.44	4.2	16.30	0.95	18.22	0.94	-11.8	12.34	2.90	9.23	1.38	25.2	0.39	0.21	0.62	0.11	-56
Feb-99	22.47	24.24	-7.8	16.03	0.64	17.44	0.66	-8.8	6.44	0.72	6.80	0.64	-5.5	0.19	0.05	0.45	0.06	-135
Mar-99	45.42	49.77	-9.6	27.57	1.08	31.35	1.15	-13.7	17.85	2.05	18.42	1.71	-3.2	0.36	0.10	0.77	0.10	-114
Apr-99	20.79	23.19	-11.6	13.38	0.50	13.94	0.48	-4.2	7.41	0.74	9.25	0.78	-24.9	0.14	0.04	0.43	0.05	-198
May-99	13.24	14.19	-7.2	8.33	0.31	7.75	0.27	7.0	4.91	0.48	6.45	0.55	-31.3	0.11	0.03	0.37	0.04	-220
Jun-99	7.64	7.94	-4.0	4.95	0.18	4.26	0.15	13.9	2.69	0.26	3.68	0.31	-36.9	0.08	0.02	0.27	0.03	-251
Jul-99	5.69	5.83	-2.5	3.85	0.14	3.27	0.11	15.0	1.84	0.17	2.56	0.21	-39.0	0.05	0.01	0.20	0.02	-303
Aug-99	6.17	6.44	-4.4	4.09	0.15	3.74	0.13	8.6	2.07	0.22	2.70	0.23	-30.1	0.04	0.01	0.17	0.02	-363
Sep-99	6.49	7.58	-16.9	4.12	0.16	4.36	0.17	-5.8	2.36	0.28	3.22	0.32	-36.2	0.03	0.01	0.17	0.02	-476
WY99 total	**176.19**	**188.53**	**-7.0**	**112.16**	**4.63**	**118.48**	**4.56**	**-5.6**	**64.03**	**8.49**	**70.04**	**6.79**	**-9.4**	**1.54**	**0.51**	**4.13**	**0.54**	**-168**
Oct-99	6.38	7.25	-13.6	4.02	0.16	4.22	0.16	-4.9	2.36	0.30	3.03	0.30	-28.3	0.03	0.01	0.18	0.02	-498
Nov-99	6.32	7.20	-14.0	4.06	0.18	4.37	0.19	-7.5	2.25	0.32	2.83	0.33	-25.6	0.04	0.02	0.21	0.03	-384
Dec-99	13.16	14.57	-10.7	8.69	0.34	9.41	0.36	-8.3	4.47	0.52	5.16	0.51	-15.4	0.12	0.03	0.42	0.05	-252
Jan-00	14.41	16.33	-13.3	10.33	0.40	11.47	0.43	-11.0	4.07	0.49	4.86	0.48	-19.2	0.12	0.04	0.39	0.05	-215
Feb-00	25.85	29.99	-16.0	16.53	0.80	19.71	0.95	-19.3	9.33	1.50	10.28	1.36	-10.2	0.19	0.07	0.47	0.08	-139
Mar-00	26.68	28.50	-6.8	18.26	0.67	19.13	0.66	-4.8	8.42	0.82	9.37	0.78	-11.2	0.15	0.04	0.42	0.05	-176
Apr-00	38.83	41.44	-6.7	21.18	0.86	22.32	0.88	-5.4	17.65	2.13	19.12	1.90	-8.3	0.27	0.08	0.66	0.09	-142
May-00	47.82	38.80	18.9	19.41	0.96	16.97	0.77	12.6	28.41	6.01	21.83	2.99	23.2	0.49	0.24	0.80	0.13	-63
Jun-00	26.67	24.71	7.4	12.98	0.56	11.18	0.46	13.9	13.69	2.01	13.53	1.58	1.2	0.30	0.10	0.65	0.09	-117
Jul-00	11.18	11.10	0.7	6.67	0.26	5.75	0.22	13.7	4.51	0.57	5.34	0.53	-18.5	0.09	0.03	0.31	0.04	-228
Aug-00	11.65	11.98	-2.9	6.93	0.27	6.47	0.24	6.7	4.72	0.58	5.52	0.52	-16.8	0.07	0.02	0.28	0.04	-286
Sep-00	8.06	7.63	5.3	4.88	0.20	4.34	0.16	11.2	3.18	0.42	3.30	0.31	-3.7	0.03	0.01	0.16	0.02	-390
WY00 total	**237.02**	**239.49**	**-1.0**	**133.95**	**5.66**	**135.34**	**5.46**	**-1.0**	**103.06**	**15.68**	**104.16**	**11.60**	**-1.1**	**1.93**	**0.68**	**4.95**	**0.70**	**-157**

Appendix. Monthly and annual inflow and outflow loads, associated errors, and removal efficiencies for selected constituents, Ellison Park wetland, Monroe County, N.Y., 1996-2001 (continued)

B. Nitrogen Compounds (continued)

Month-year	Total nitrogen			Nitrate plus nitrite					Ammonia-plus-organic nitrogen					Ammonia nitrogen				
	Inflow	Outflow		Inflow		Outflow			Inflow		Outflow			Inflow		Outflow		
	Load	Load	RE	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE
Oct-00	7.64	7.32	4.2	4.76	0.18	4.24	0.15	10.9	2.88	0.31	3.08	0.26	-6.9	0.03	0.01	0.17	0.02	-449
Nov-00	8.33	8.35	-0.2	5.22	0.22	5.02	0.20	4.0	3.11	0.41	3.33	0.33	-7.2	0.05	0.02	0.23	0.03	-347
Dec-00	15.28	17.56	-14.9	9.63	0.41	10.76	0.48	-11.7	5.65	0.82	6.81	0.89	-20.4	0.12	0.04	0.45	0.07	-262
Jan-01	13.89	14.53	-4.6	10.16	0.42	10.29	0.41	-1.3	3.73	0.47	4.24	0.46	-13.7	0.09	0.03	0.30	0.04	-227
Feb-01	30.76	32.77	-6.5	20.92	0.84	22.48	0.85	-7.5	9.85	1.11	10.29	0.99	-4.5	0.20	0.05	0.50	0.07	-154
Mar-01	52.60	53.06	-0.9	30.31	1.25	31.80	1.22	-4.9	22.29	2.77	21.26	2.10	4.6	0.30	0.09	0.68	0.09	-124
Apr-01	30.21	31.53	-4.4	18.04	0.73	17.93	0.68	0.6	12.16	1.35	13.60	1.26	-11.9	0.16	0.04	0.46	0.06	-188
May-01	12.61	12.80	-1.5	7.93	0.30	6.76	0.24	14.9	4.68	0.49	6.04	0.53	-29.2	0.08	0.02	0.28	0.03	-266
Jun-01	7.94	8.44	-6.3	5.11	0.20	4.33	0.16	15.1	2.83	0.30	4.10	0.36	-44.8	0.06	0.01	0.23	0.03	-320
Jul-01	5.36	5.45	-1.8	3.65	0.14	2.97	0.11	18.4	1.71	0.17	2.48	0.21	-45.0	0.03	0.01	0.16	0.02	-390
Aug-01	5.35	5.35	-0.1	3.58	0.15	3.04	0.12	15.2	1.76	0.21	2.31	0.22	-31.2	0.02	0.01	0.12	0.02	-459
Sep-01	7.26	7.25	0.3	4.22	0.22	3.90	0.20	7.5	3.05	0.62	3.34	0.46	-9.7	0.03	0.01	0.14	0.02	-432
WY01 total	**197.23**	**204.41**	**-3.6**	**123.53**	**5.07**	**123.51**	**4.81**	**0.0**	**73.70**	**9.03**	**80.90**	**8.06**	**-9.8**	**1.17**	**0.34**	**3.73**	**0.50**	**-218**
Period of record WY97-01	**1,148.93**	**1,149.81**	**-0.1**	**695.71**	**30.30**	**699.57**	**28.45**	**-0.6**	**453.22**	**64.17**	**450.24**	**47.27**	**0.7**	**10.37**	**3.57**	**26.23**	**3.80**	**-153**

Appendix. Monthly and annual inflow and outflow loads, associated errors, and removal efficiencies for selected constituents, Ellison Park wetland, Monroe County, N.Y., 1996-2001 (continued)

[Inflow and outflow locations refer to Irondequoit Creek above Blossom Road and at Empire Boulevard, respectively. Load and error values are in tons. Error is standard error of prediction, which, when multiplied by 1.96 and added to and subtracted from the estimated load, provides 95-percent confidence limits of the load estimate. RE, removal efficiency, in percent, is computed from load values prior to rounding. Water year REs are based on total annual loads, not mean monthly values. Positive (negative) RE value indicates net retention in (export from) wetland, respectively. WY, water year. See Coon and others (2000) for 1990-96 data.]

C. Suspended Solids

Month-year	Total suspended solids					Volatile suspended solids				
	Inflow		Outflow			Inflow		Outflow		
	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE
Oct-96	5,018	2,616	2,074	641	58.7	540	219	248	53	54.0
Nov-96	2,841	1,109	1,520	383	46.5	359	110	211	38	41.4
Dec-96	4,027	1,472	2,058	493	48.9	488	142	274	48	43.9
Jan-97	978	282	747	164	23.6	144	36	116	20	19.2
Feb-97	1,774	641	1,177	299	33.6	219	65	150	28	31.8
Mar-97	2,723	967	968	327	64.4	290	88	140	44	51.7
Apr-97	1,361	513	598	204	56.1	156	49	87	27	44.0
May-97	1,013	465	462	165	54.4	120	43	67	21	44.0
Jun-97	958	367	491	159	48.7	119	37	73	21	39.2
Jul-97	393	148	262	84	33.3	57	17	43	12	24.8
Aug-97	664	355	404	142	39.2	88	35	65	20	26.2
Sep-97	2,471	2,003	796	350	67.8	255	156	119	43	53.3
WY97 total	**24,221**	**10,938**	**11,557**	**3,410**	**52.3**	**2,835**	**998**	**1,593**	**378**	**43.8**
Oct-97	704	295	492	147	30.1	96	32	84	22	12.8
Nov-97	1,537	608	838	245	45.5	189	62	135	35	28.7
Dec-97	1,096	343	763	193	30.4	142	38	123	29	12.9
Jan-98	21,707	15,314	4,725	2,322	78.2	1,633	914	546	222	66.6
Feb-98	2,218	848	975	272	56.0	240	73	134	32	44.3
Mar-98	3,846	1,216	1,897	466	50.7	399	103	237	51	40.6
Apr-98	1,000	314	706	171	29.3	122	32	95	20	22.6
May-98	683	283	488	141	28.5	87	28	66	16	23.5
Jun-98	1,633	967	875	319	46.4	179	82	110	33	38.7
Jul-98	11,934	8,295	4,116	1,907	65.5	1,034	550	439	163	57.5
Aug-98	2,251	1,449	1,100	395	51.1	254	121	150	44	40.9
Sep-98	1,071	403	775	212	27.6	144	44	117	28	18.6
WY98 total	**49,679**	**30,335**	**17,750**	**6,789**	**64.3**	**4,521**	**2,078**	**2,237**	**696**	**50.5**
Oct-98	651	279	523	148	19.7	92	31	85	21	7.5
Nov-98	227	69	255	65	-12.3	37	10	46	11	-22.4
Dec-98	196	72	233	62	-18.7	31	9	41	10	-30.8
Jan-99	3,434	2,750	1,259	560	63.3	311	194	158	58	49.4
Feb-99	867	288	736	191	15.1	109	30	103	23	5.4
Mar-99	3,874	1,397	2,622	682	32.3	401	116	305	68	24.1
Apr-99	1,094	337	1,055	248	3.5	136	34	133	28	2.2
May-99	648	196	690	170	-6.5	88	22	91	20	-3.0
Jun-99	361	113	425	109	-17.5	54	14	60	14	-11.0
Jul-99	290	83	362	90	-24.6	47	11	54	12	-17.0
Aug-99	548	206	562	151	-2.6	81	24	85	20	-4.3
Sep-99	747	309	766	238	-2.5	106	34	115	30	-9.1
WY99 total	**12,938**	**6,102**	**9,487**	**2,715**	**26.7**	**1,493**	**530**	**1,274**	**314**	**14.6**

Appendix. Monthly and annual inflow and outflow loads, associated errors, and removal efficiencies for selected constituents, Ellison Park wetland, Monroe County, N.Y., 1996-2001 (continued)

C. Suspended Solids (continued)

Month-year	Total suspended solids					Volatile suspended solids				
	Inflow		Outflow			Inflow		Outflow		
	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE
Oct-99	578	281	551	173	4.6	83	30	89	24	-6.9
Nov-99	328	159	341	121	-3.9	48	18	56	17	-17.8
Dec-99	527	207	514	157	2.4	74	23	82	21	-10.8
Jan-00	400	168	425	128	-6.3	56	18	67	17	-19.3
Feb-00	1,665	834	1,285	478	22.8	177	71	158	49	11.0
Mar-00	1,044	285	937	208	10.3	133	31	127	25	4.3
Apr-00	3,418	1,306	2,355	645	31.1	363	110	274	64	24.4
May-00	9,720	7,467	3,088	1,274	68.2	824	480	334	111	59.5
Jun-00	3,502	1,768	1,910	663	45.4	375	145	230	65	38.9
Jul-00	1,033	473	801	244	22.4	135	47	113	29	16.6
Aug-00	1,499	643	1,090	302	27.3	195	64	159	37	18.5
Sep-00	993	472	591	163	40.5	134	48	97	23	27.4
WY00 total	**24,706**	**14,062**	**13,888**	**4,556**	**43.8**	**2,597**	**1,087**	**1,785**	**483**	**31.3**
Oct-00	596	210	411	105	31.0	89	25	73	17	17.4
Nov-00	405	171	292	84	28.0	59	20	54	13	9.9
Dec-00	690	360	578	233	16.2	90	36	93	30	-3.0
Jan-01	297	138	258	88	13.1	43	15	45	12	-4.9
Feb-01	1,109	386	768	210	30.8	138	39	116	27	15.4
Mar-01	3,901	1,559	1,890	534	51.5	406	130	249	60	38.6
Apr-01	1,630	546	1,024	269	37.2	192	54	143	33	25.9
May-01	443	149	354	95	19.9	63	18	56	13	11.7
Jun-01	272	95	256	72	5.9	42	12	43	11	-1.5
Jul-01	192	63	182	51	5.0	32	9	34	9	-3.9
Aug-01	335	145	235	73	29.8	52	18	44	12	14.5
Sep-01	1,012	797	420	193	58.5	124	73	74	28	39.9
WY01 total	**10,881**	**4,620**	**6,668**	**2,007**	**38.7**	**1,330**	**450**	**1,024**	**266**	**23.1**
Period of record WY97-01	**122,425**	**66,058**	**59,351**	**19,477**	**51.5**	**12,776**	**5,144**	**7,912**	**2,137**	**38.1**

Appendix. Monthly and annual inflow and outflow loads, associated errors, and removal efficiencies for selected constituents, Ellison Park wetland, Monroe County, N.Y., 1996-2001

[Inflow and outflow locations refer to Irondequoit Creek above Blossom Road and at Empire Boulevard, respectively. Load and error values are in tons. Error is standard error of prediction, which, when multiplied by 1.96 and added to and subtracted from the estimated load, provides 95-percent confidence limits of the load estimate. RE, removal efficiency, in percent, is computed from load values prior to rounding. Water year REs are based on total annual loads, not mean monthly values. Positive (negative) RE value indicates net retention in (export from) wetland, respectively. WY, water year. See Coon and others (2000) for 1990-96 data.]

D. Selected Anions (chloride and sulfate)

Month-year	Chloride					Sulfate				
	Inflow		Outflow			Inflow		Outflow		
	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE
Oct-96	1,321	71	1,395	72	-5.6	1,493	61	1,484	55	0.6
Nov-96	1,655	73	1,751	78	-5.8	1,597	59	1,572	55	1.6
Dec-96	2,741	117	2,847	124	-3.8	1,959	72	1,896	66	3.2
Jan-97	1,971	81	2,229	94	-13.1	1,298	47	1,316	45	-1.4
Feb-97	2,381	109	2,648	122	-11.2	1,370	54	1,376	50	-0.4
Mar-97	3,105	138	3,312	143	-6.7	1,926	66	1,896	61	1.6
Apr-97	1,699	79	1,820	83	-7.2	1,413	48	1,402	45	0.8
May-97	986	45	1,026	46	-4.1	1,137	37	1,102	34	3.1
Jun-97	752	35	806	37	-7.1	1,005	33	996	31	0.9
Jul-97	442	20	468	21	-6.0	753	25	727	22	3.4
Aug-97	442	21	494	23	-11.6	766	25	776	24	-1.4
Sep-97	588	33	627	33	-6.7	906	32	908	29	-0.3
WY97 total	**18,083**	**825**	**19,423**	**876**	**-7.4**	**15,623**	**559**	**15,453**	**516**	**1.1**
Oct-97	585	27	693	31	-18.5	895	28	961	28	-7.3
Nov-97	1,153	55	1,302	61	-12.9	1,229	39	1,284	38	-4.4
Dec-97	1,718	78	2,050	90	-19.3	1,376	42	1,488	42	-8.2
Jan-98	4,616	270	4,882	283	-5.8	2,501	100	2,394	87	4.3
Feb-98	3,141	149	3,127	145	0.5	1,693	55	1,614	49	4.7
Mar-98	3,727	165	3,972	171	-6.6	2,168	68	2,196	64	-1.3
Apr-98	1,625	72	1,739	75	-7.0	1,354	42	1,388	40	-2.5
May-98	935	43	962	43	-2.9	1,071	33	1,056	30	1.4
Jun-98	757	39	820	40	-8.2	988	32	1,013	31	-2.5
Jul-98	1,362	76	1,422	78	-4.4	1,544	54	1,559	50	-1.0
Aug-98	727	36	745	35	-2.5	1,031	33	1,037	31	-0.6
Sep-98	647	30	680	30	-5.0	964	30	988	29	-2.5
WY98 total	**20,994**	**1,041**	**22,396**	**1,081**	**-6.7**	**16,813**	**554**	**16,977**	**517**	**-1.0**
Oct-98	622	28	673	29	-8.1	920	28	950	27	-3.2
Nov-98	622	28	696	30	-11.9	812	25	836	24	-2.9
Dec-98	894	40	1,038	45	-16.1	870	27	912	26	-4.8
Jan-99	2,305	146	2,595	148	-12.6	1,355	52	1,409	48	-4.0
Feb-99	2,458	115	2,707	124	-10.2	1,372	44	1,433	43	-4.4
Mar-99	3,845	174	4,354	192	-13.3	2,173	70	2,336	71	-7.5
Apr-99	1,872	81	2,111	89	-12.7	1,470	46	1,590	46	-8.1
May-99	1,086	47	1,194	50	-10.0	1,165	35	1,225	35	-5.2
Jun-99	619	27	679	29	-9.6	861	27	891	26	-3.4
Jul-99	492	21	537	22	-9.2	780	24	799	23	-2.5
Aug-99	549	24	598	25	-8.8	850	26	885	25	-4.1
Sep-99	599	28	687	31	-14.6	895	28	973	29	-8.8
WY99 total	**15,964**	**759**	**17,869**	**815**	**-11.9**	**13,523**	**433**	**14,238**	**422**	**-5.3**

Appendix. Monthly and annual inflow and outflow loads, associated errors, and removal efficiencies for selected constituents, Ellison Park wetland, Monroe County, N.Y., 1996-2001 (continued)

D. Selected Anions (chloride and sulfate) (continued)

Month-year	Chloride					Sulfate				
	Inflow		Outflow			Inflow		Outflow		
	Load	Error	Load	Error	RE	Load	Error	Load	Error	RE
Oct-99	636	29	707	31	-11.1	911	28	959	28	-5.2
Nov-99	670	34	756	38	-12.8	809	26	839	26	-3.7
Dec-99	1,399	64	1,560	70	-11.5	1,141	35	1,185	34	-3.9
Jan-00	1,727	79	1,983	89	-14.8	1,135	35	1,200	35	-5.8
Feb-00	2,576	141	3,023	166	-17.4	1,389	49	1,497	51	-7.7
Mar-00	2,867	126	3,111	134	-8.5	1,672	51	1,742	50	-4.2
Apr-00	2,871	137	3,180	150	-10.7	1,956	64	2,061	64	-5.4
May-00	2,259	123	2,266	116	-0.3	1,925	68	1,872	59	2.7
Jun-00	1,458	72	1,510	72	-3.6	1,483	48	1,500	45	-1.1
Jul-00	815	38	872	39	-7.0	1,038	32	1,068	31	-2.9
Aug-00	890	41	966	43	-8.5	1,125	34	1,184	34	-5.2
Sep-00	720	34	724	32	-0.6	978	31	973	28	0.6
WY00 total	**18,889**	**919**	**20,658**	**982**	**-9.4**	**15,562**	**502**	**16,079**	**484**	**-3.3**
Oct-00	765	34	756	32	1.2	1,000	30	969	27	3.1
Nov-00	865	43	904	43	-4.5	929	29	917	27	1.3
Dec-00	1,559	77	1,781	90	-14.2	1,183	37	1,224	37	-3.5
Jan-01	1,756	85	1,916	91	-9.1	1,094	34	1,095	32	-0.1
Feb-01	3,343	159	3,690	172	-10.4	1,619	54	1,667	52	-2.9
Mar-01	4,445	214	4,835	224	-8.8	2,285	78	2,324	74	-1.7
Apr-01	2,595	125	2,830	132	-9.1	1,710	57	1,758	55	-2.8
May-01	1,100	49	1,177	51	-7.0	1,112	36	1,115	33	-0.2
Jun-01	674	31	757	33	-12.3	856	28	873	27	-2.0
Jul-01	494	22	548	24	-10.9	738	25	737	23	0.2
Aug-01	512	24	553	25	-8.1	764	27	757	24	0.8
Sep-01	624	35	665	36	-6.6	848	32	837	29	1.2
WY01 total	**18,732**	**899**	**20,413**	**953**	**-9.0**	**14,139**	**468**	**14,273**	**441**	**-0.9**
Period of record WY97-01	**92,662**	**4,443**	**100,759**	**4,707**	**-8.7**	**75,659**	**2,515**	**77,020**	**2,380**	**-1.8**