Sources of fecal coliform bacteria and plant nutrients and their effect on water quality in Swan Lake, Carroll County, Iowa

by

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INTRODUCTION

Swan Lake is located in west-central Iowa in Carroll County. The lake lies 4 km south of the town of Carroll and approximately 145 km northwest of Des Moines. The location of Swan Lake within the state and Carroll County is shown in Figures 1 and 2. The impoundment was constructed in 1935 on an unnamed tributary near the headwaters of the Middle Raccoon River. The lake and surrounding park are owned by the state and managed by the Carroll County Conservation Board for multiple-use recreation which includes picnicking, camping, swimming, and fishing. The park is one of the most intensively developed multi-use areas in Iowa (Bachmann et al. 1982). There are few lakes available for recreation in this part of Iowa, so Swan Lake is a focus for water-based outdoor recreation. The park and associated lake has an annual use estimated at over 400,000 visitations (IDNR 1993). According to the Iowa water quality standards (IAC 1990) , Swan Lake is designated for primary contact recreation (Class A uses) and for aquatic life uses typical of lakes and wetlands (Class B (LW) uses). The water quality standards contain numeric criteria for Class A and Class B (LW) waters to ensure that water quality supports these uses.

The most recent physical measurements of the lake were taken from a 1990 map. The lake has a surface area of 45.5 ha and a watershed consisting of 351.2 ha. The watershed to lake area ratio is approximately 8:1. Swan Lake has a maximum depth of 4.3 m and a mean depth of 1.7 m. Swan Lake State Park occupies approximately 243 ha of the watershed. The watershed within the park consists of woods, meadow, and cut-grass areas. The watershed which lies outside the park is used primarily for agricultural purposes including crop and cattle production. The lake is fed by several small intermittent tributaries and tile lines found throughout the watershed.

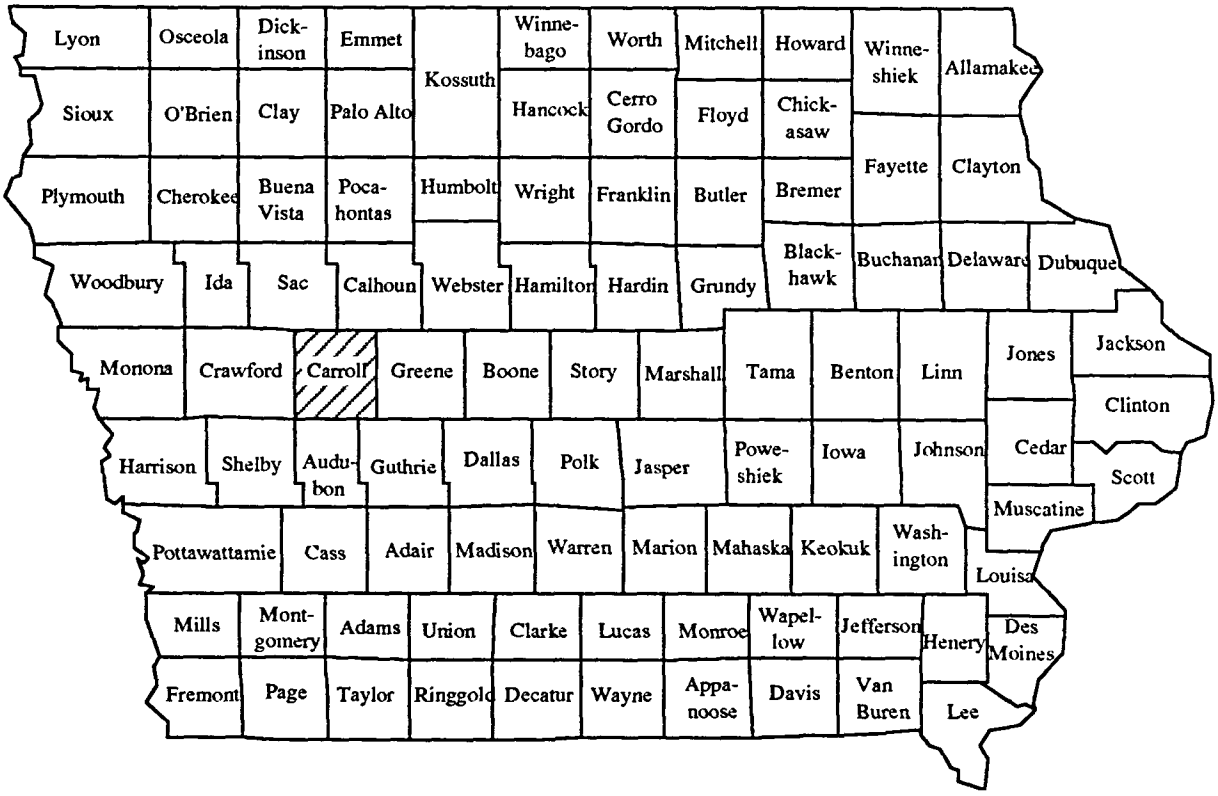

Figure 1. Location of Swan Lake within the state of Iowa

History of Swan Lake

Prior to any water quality studies conducted on Swan Lake there was recognition of water quality problems due to a combination of factors. The first problem was construction design. The lake was constructed with a maximum depth of only 2 m. This depth was not adequate to ensure oxygen levels necessary to over-winter fish and as a result the lake experienced four fish kills between 1970 and 1980 (Bachmann et al. 1982). Second, siltation of the shallow lake basin required raising the lake level by 0.3 m in 1971. The third problem resulted from an intensive effort to decrease soil erosion in the watershed. The implementation of soil conservation practices resulted in decreased runoff entering the lake. The combination

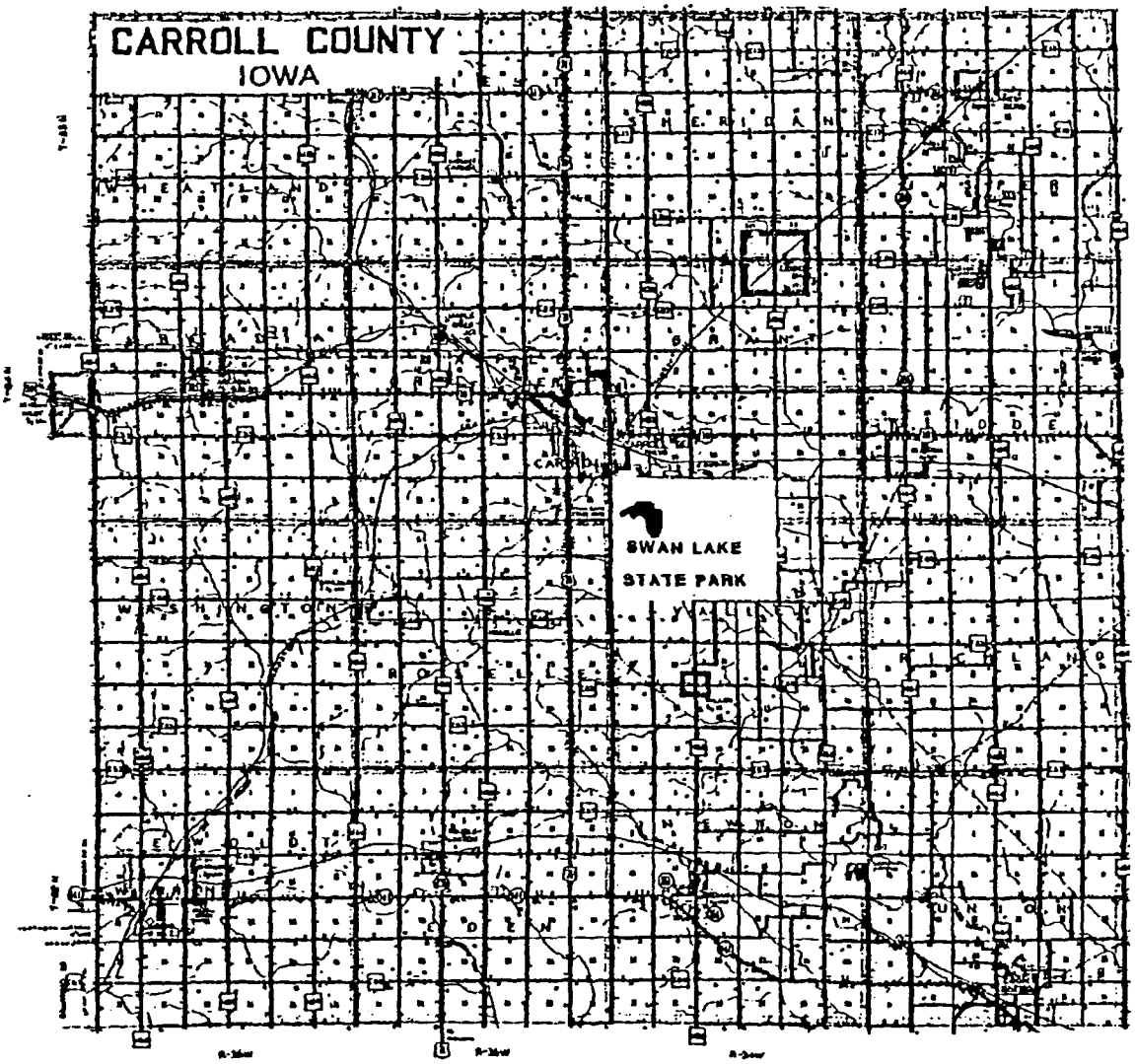

Figure 2. Location of Swan Lake within Carroll County

of a shallow lake, siltation, and lowering lake levels created poor water quality and decreased use of an intensively managed multi-use recreational area.

The Iowa Clean Lakes Study of 1979 (Bachmann et al. 1980) classified Iowa's lakes according to lake water quality. A total of 107 lakes were surveyed and Swan Lake ranked fifth on the priority list for restoration. A diagnostic feasibility study in 1980-1981 (Bachmann

et al. 1982) identified low levels of dissolved oxygen during the winter months, low water transparency, and high levels of plant nutrients leading to nuisance algal blooms as the major water quality problems of Swan Lake. Recommendations for restoration were made and the following activities for restoration were conducted from 1982-1986 as part of the Clean Lakes Phase II Program at Swan Lake.

Draining of Swan Lake began in September 1982 and was completed in November 1982. Construction of tile outlet terraces on agricultural land bordering the watershed began in fall of 1982 and was completed in summer 1983. Water draining from this land is now delivered to the lake by a tile line. This work improved the water budget by adding approximately 36 ha to the watershed of Swan Lake.

Ten jetties were constructed in Swan Lake during the summer of 1983. Approximately 15,300 cubic meters of sediment were pushed from the lake bottom to form these structures. The jetties deepened shoreline areas, improved fish habitat, and provided access for anglers.

Excavation of Swan Lake took place in the winter of 1983-1984. Approximately 102,000 cubic meters of sediment were removed from the lower portion of the lake and deposited in the upper end of the lake to increase marsh area. The maximum depth of the lake was increased from 2 m to 4.5 m. Lake surface area was reduced from 52.6 ha to 45.5 ha to improve the water budget of Swan Lake.

During the period September 1984 to February 1985 six-thousand tons of fieldstone were placed around the shoreline. The rip-rap reduced bank erosion and improved fish habitat. Prior to refilling the lake in 1985, additional fish habitat was placed on the lake bottom. This included stake beds, cribs, and brush piles.

Fish renovation and restocking began with the filling of the lake in the summer of 1985. Rotenone was used to remove fish from the lake prior to refilling. Swan

Lake was restocked with game fish which included largemouth bass (*Micropterus salmoides*), channel catfish (*Ictalurus punctatus*), and bluegill (*Lepomis macrochirus*).

Aeration equipment was installed in the period September 1985 to February 1986. The aeration system consisted of two blowers housed on shore and four helixors placed in the deepest portion of the lake.

A well was drilled in the spring of 1985. The capacity of the well was 400 gallons per minute and was developed to maintain the lake level during drought conditions. The pipeline to deliver water to the lake was completed in the spring of 1986.

During the years 1987-1989 a post-restoration study of Swan Lake was conducted to determine if restoration had improved lake water quality (IDNR 1993). Restoration practices resulted in some improvements to Swan Lake and increased the recreational value of the lake and surrounding park. By increasing the maximum depth of the lake and installing aeration equipment low dissolved oxygen was no longer a problem, and fish kills have not occurred. The addition of land to the watershed and a well for use during dry periods has increased the lake's water supply and has helped stabilize lake levels.

Although restoration improved the recreational value of the lake by reducing the risk of fish kills and stabilizing lake levels, it was found that water quality had not appreciably improved. The water quality monitoring program that was conducted from 1987-1989 showed relatively high levels of fecal coliform bacteria at the east swimming beach during the summer months (IDNR 1993). These levels exceeded the Iowa water quality standard (200 fecal coliform colonies per 100 ml) for protection of primary contact uses in 8 of 22 samples. In addition to fecal coliform bacteria, Swan Lake still suffered from high levels of nutrients that caused nuisance algal blooms and reduced water transparency (see Appendices A and B for discussions on bacterial indicators and nutrients).

The 1987-1989 post-restoration study (IDNR 1993) suggested improper livestock waste management practices in the watershed were resulting in water quality deterioration. The application of livestock waste to fields in the watershed during the winter months may have been contributing high concentrations of bacteria and nutrients to the lake through runoff derived from snow melt and rainfall. In addition, there is a cattle feedlot and lagoons located in the watershed not far from the lake. There was some concern the lagoons were overflowing during storm events. Swan Lake State Park also has two wildlife pens that house wildlife such as buffalo, whitetail deer, and wild turkeys. Runoff from the pens may also contribute bacteria and nutrients to the lake.

Study Objectives

The specific objectives of this study were to: (1) identify the major sources of fecal coliform bacteria and plant nutrients from the surrounding watershed, (2) monitor lake water quality, (3) determine the importance of inputs of bacteria and nutrients from the watershed to the water quality problems in Swan Lake, and (4) make recommendations to improve the water quality of Swan Lake.

MATERIALS AND METHODS

Study Design

To determine the major sources and inputs of bacteria and nutrients from the lake's watershed two types of sampling schemes were required. Routine sampling involved regular monitoring of the lake and flowing inlets, tiles, and wildlife pen runoff that enters the lake. In addition to routine sampling, special runoff events were sampled to monitor the flowing inlets, tiles, and wildlife pen runoff. The criterion used to define a runoff event was when at least 2.5 cm (1 inch) of rain fell in a 24-hour period. This criterion allowed for sampling of the heavier runoff events that may have a greater impact on water quality than non-runoff inputs to the lake. This design allowed for comparisons between sites and between non-runoff and runoff conditions.

Possible sources of fecal coliform bacteria and nutrients included nonpoint runoff from agricultural land and from specific sources in the watershed such as feedlots, lagoons, and wildlife pens in the park. Ten sampling sites were established in the watershed to examine these sources. In addition, five sampling sites were established in the lake to monitor water quality. Samples were collected from the following locations (Figure 3):

Lake sites

L1	deepest portion of lake
L2	mid-lake
L3	upper end of lake
L4	west swimming beach
L5	east swimming beach

Inlets

INW	inlet entering lake from the west
INCE	tile draining field to the southwest that pours into INW
INCW	inlet above INCE
INSW	inlet entering lake from the southwest
INWP3	runoff from wildlife pen number three
INS	tile draining southern part of watershed
INSE	inlet draining southeast part of watershed
INWP1	runoff from wildlife pen number one
INN1	inlet draining northern part of watershed
INN2	tile draining northern part of watershed

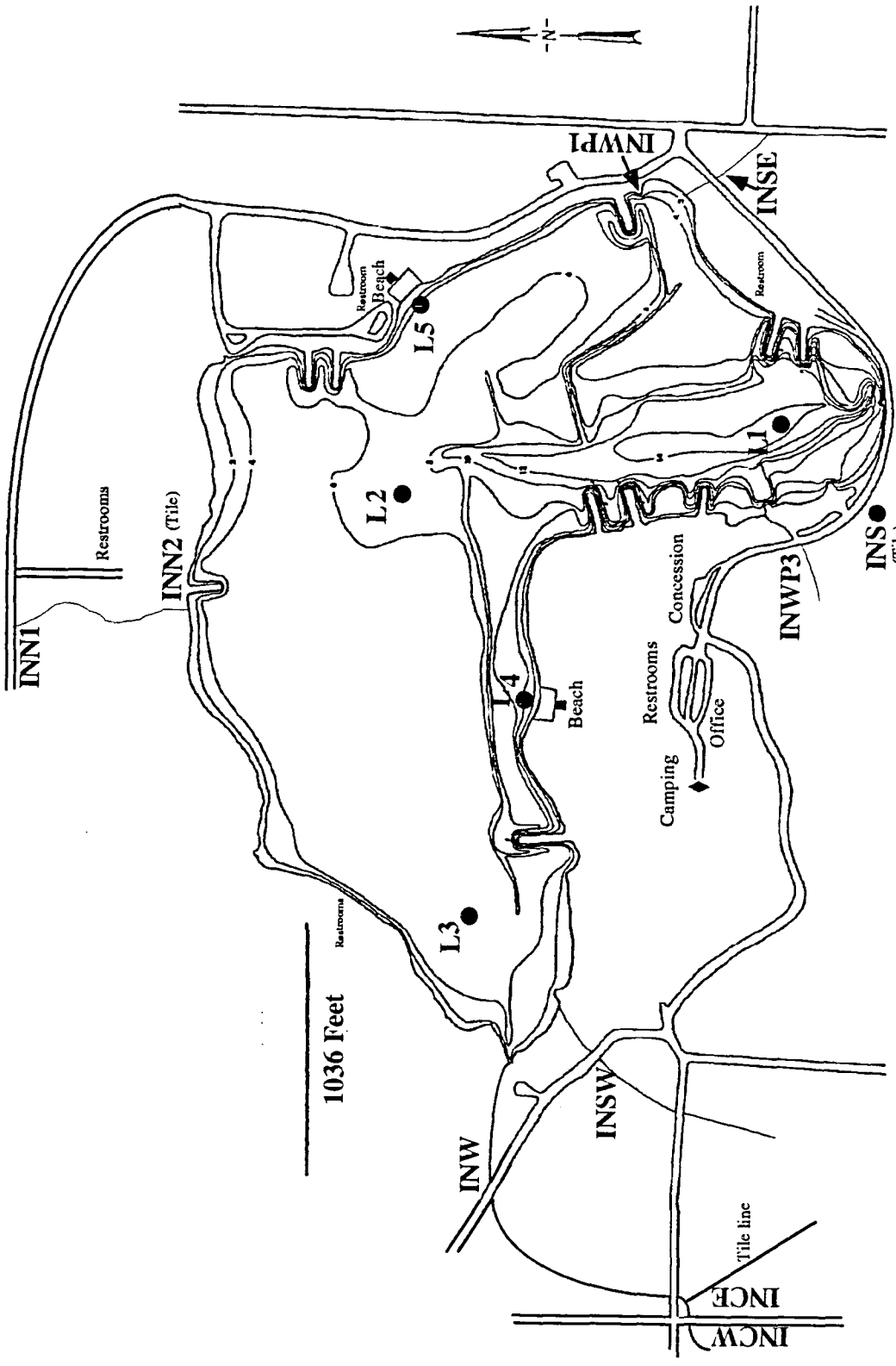

Figure 3. Map of Swan Lake showing locations of sampling sites in 1993

Site L1 was located at the deepest portion of the lake (approximately 4 m) near the dam. Samples at this site were collected at depths of 0.5 m, 2.0 m, and 3.5 m below the surface. Samples were collected at 0.5 m below the surface at lake sites L2 and L3. Samples collected at Sites L4 and L5 were taken over approximately 1 m of water just below the surface. These five sites were selected to show temporal and spatial changes in water quality throughout the lake.

Figure 4 shows the inlets, tiles, and wildlife pen runoff in relation to the watershed. Site INW is on the main inlet entering the lake from the west. This site is located next to the lake and would include nonpoint runoff from the surrounding agricultural land. Site INCE is located on the southwest portion of the watershed. This site is a tile that drains a corn field and may also include runoff from a bordering cattle operation. This tile discharges into the main west inlet. Site INCW is also located on the main west inlet but above the tile site INCE. This site comprises nonpoint runoff from agricultural land. Site INSW is located on the inlet entering the lake from the southwest. This site includes possible drainage from lagoons that contain cattle feedlot runoff. Site INWP3 is located next to the lake and includes runoff from wildlife pen number three. Site INS is a tile that discharges into the lake and drains 36 ha of agricultural land added during restoration on the southern part of the watershed. Site INSE is located on an inlet draining agricultural land on the southeast part of the watershed. Site INWP1 is located next to the lake and contains runoff from wildlife pen number one. Site INN1 is located on an inlet that drains mostly land in the park and a small amount of cropland on the northern part of the watershed. Site INN2 is a tile that discharges into the lake, and also drains land in the park on the northern part of the watershed. These ten sites were selected to determine the importance of inputs from the watershed.

In addition to sampling Swan Lake inlets, tiles, and wildlife pen runoff, a control stream was selected that was not potentially affected by animal waste and sampled for fecal coliform bacteria. Data from this site were compared to Swan Lake inlets to determine if Swan

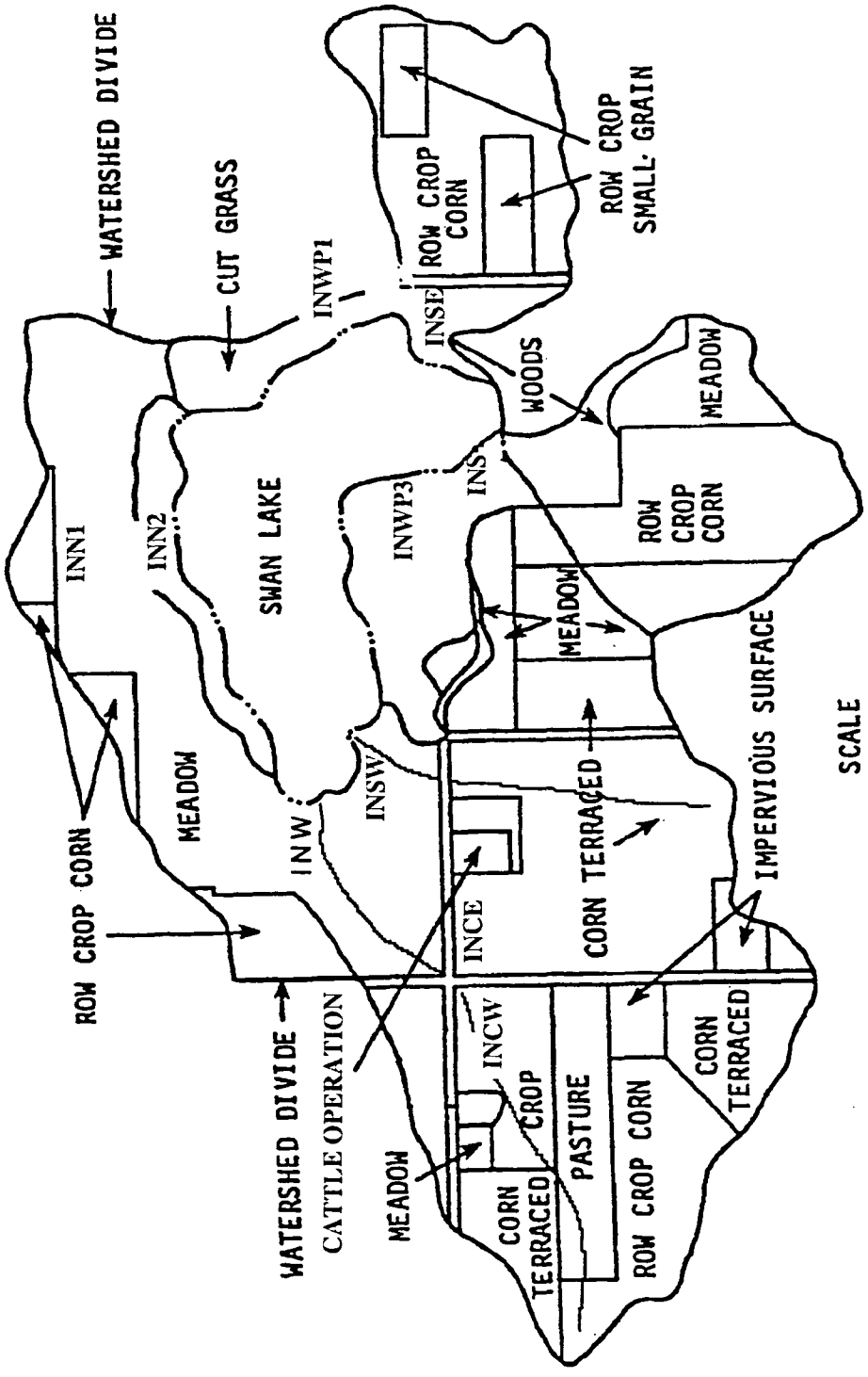

Figure 4. Map showing sampling sites in the Swan Lake watershed in 1993. Map of watershed and land use practices was taken from Bachmann et al. (1982)

Lake inlets were transporting unusually high concentrations of bacteria.

Water samples were collected from March 10, 1993 to October 12, 1993. Table 1 lists the parameters analyzed, frequency of sampling, and sampling locations for routine monitoring and runoff events. Routine sampling was conducted bi-weekly after spring ice-out. Because the inlets, tiles, and wildlife pens were not always flowing, only those that were flowing during the time of sample collection were analyzed.

Sample Analysis

The Iowa State University Limnology Laboratory performed all sample analysis and collected samples during routine monitoring. Samples were collected between 0800 and 1600 hours except those collected during the special runoff events. Volunteers from a local chapter of the Iowa Conservation Education Council (ICEC) assisted in sample collection during runoff events. Volunteers were trained during an actual sample collection on June 9, 1993. Volunteers were taken to each sampling site in the watershed and instructed on techniques used for collection. After samples were collected for runoff events volunteers placed all samples on ice and contacted Iowa State University researchers to arrange for sample transport. Samples were received, transported, and analyzed on the same day as sample collection.

Samples were analyzed according to USEPA approved methods. A quality assurance workplan was submitted to the EPA and IDNR and approved (Bachmann and Hoyman 1993). Standard operating procedures were written and rigorously followed for each test.

Measurements taken at lake sites included vertical temperature and dissolved oxygen profiles. Temperature and dissolved oxygen were determined using an air calibrated, YSI Model 50b dissolved oxygen probe. Secchi disk transparencies were also taken at each lake site using a standard alternating black and white quadrant, 20 cm diameter Secchi disk. Water samples were collected with a Van dorn sampler. All water was transferred from the Van dorn sampler directly into acid washed Nalagene bottles and stored on ice. Prior to filling the Nalagene bottles they were rinsed at each site with lake water. At each site three 1 L bottles

Table 1. Schedule of water quality analysis performed on samples taken from Swan Lake in 1993

Samples taken on routine sample runs

Parameter	Frequency [a]	Sites
Algae biovolume	May - September	L1
Secchi disk depth	April - October	L1,L2, & L3
Temperature	April - October	L1,L2, & L3
Dissolved oxygen	April - October	L1,L2, & L3
pH	April - October	L1,L2, & L3
Total Alkalinity	April - October	L1,L2, & L3
Chlorophyll a	April - October	L1,L2, & L3
Total phosphorus	April - October	L1,L2,L3,Inlets,tiles,&pens
Soluble reactive phosphorus	April - October	L1,L2,L3,Inlets,tiles,&pens
Total suspended solids	April - October	L1,L2,L3,Inlets,tiles,&pens
Inorganic suspended solids	April - October	L1,L2,L3,Inlets,tiles,&pens
Organic suspended solids	April - October	L1,L2,L3,Inlets,tiles,&pens
Total nitrogen	April - October	L1,L2,L3,Inlets,tiles,&pens
Nitrate nitrogen	April - October	L1,L2,L3,Inlets,tiles,&pens
Ammonia nitrogen	April - October	L1,L2,L3,Inlets,tiles,&pens
Organic nitrogen	April - October	L1,L2,L3,Inlets,tiles,&pens
Fecal coliforms	June - October	L1,L2,L3,L4,L5, Inlets, tiles, & pens

Samples taken on runoff events

Parameter	Frequency [b]	Sites
Total phosphorus	March - October	Inlets,tiles, & pens
Soluble reactive phosphorus	March - October	Inlets,tiles, & pens
Total suspended solids	March - October	Inlets,tiles, & pens
Inorganic suspended solids	March - October	Inlets,tiles, & pens
Organic suspended solids	March - October	Inlets,tiles, & pens
Total nitrogen	March - October	Inlets,tiles, & pens
Nitrate nitrogen	March - October	Inlets,tiles, & pens
Ammonia nitrogen	March - October	Inlets,tiles, & pens
Organic nitrogen	March - October	Inlets,tiles, & pens
Fecal coliforms	June - October	Inlets,tiles, & pens

[a] bi-weekly.

[b] irregular depending on the timing of rain events.

were filled for chlorophyll a, suspended solids, and algal biovolume (site L1 only) analysis. In addition, one 500 ml bottle was filled for phosphorus, pH, and total alkalinity analysis. One 250 ml bottle was also filled for nitrogen analysis. Nitrogen samples were preserved in the field using 0.5 ml of concentrated sulfuric acid for each 250 ml bottle of water sample. One 250 ml sterilized bottle was submerged by hand to approximately 0.5 m and filled for fecal coliform bacteria analysis. At one of the sites duplicate sets of samples were taken from the same depth. The pH was measured on shore immediately after sampling with an Orion Research Ionalyzer Model 250A calibrated with commercial pH buffers.

At each sampling site in the watershed acid washed Nalagene bottles were used to collect water samples for suspended solids, phosphorus, and nitrogen. One 1 L bottle was filled for suspended solids analysis and one 500 ml bottle for phosphorus analysis. A 250 ml bottle was filled with water and preserved with 0.5 ml concentrated sulfuric acid for nitrogen analysis. In addition two (duplicate) sterilized 250 ml bottles were filled for fecal coliform bacteria analysis. All bottles, except bacteria bottles, were rinsed prior to filling at each site. Bottles were filled by hand, submerging them below the surface, and then placed on ice for transport.

Immediately after returning from the field soluble reactive phosphorus samples were filtered through Gelman 0.45 µm membrane filters. The filtrate, which contained the dissolved phosphorus, was treated with an ascorbic acid solution (APHA et al. 1989). Duplicate phosphorus standards and blanks were also treated with the ascorbic acid solution and the absorbance at 880 nm was recorded spectrophotometrically on a Gilford Response II to produce a concentration curve on each day of analysis. Absorbance values of the samples were then determined and their concentrations calculated from the standard concentration curve. Phosphorus standard solutions were prepared from a 1000 mg/l P stock solution available commercially.

Total phosphorus concentration was determined after ammonium persulfate digestion of phosphorus standards, blanks, and lake water samples in an autoclave at 18 psi for 40 minutes to convert all forms of phosphorus to orthophosphate (APHA et al. 1989). Phosphorus standards, blanks, and lake water samples were treated with an ascorbic acid solution (APHA et al. 1989). The absorbance of duplicate standards and blanks at 880 nm was recorded spectrophotometrically on a Gilford Response II to produce a concentration curve on each day of analysis. Absorbance values of the samples were then determined and concentrations calculated from the standard concentration curve.

Total alkalinity, measured as $CaCO_3$, was determined following the buret titration method of Hach Company (1989). Samples were titrated with standardized 0.020 N sulfuric acid until reaching a pH of 4.8. The pH was measured with an Orion Research Ionalyzer Model 250A calibrated with commercial pH buffers.

Suspended solid filters were combusted at 550 °C in a muffle furnace and weighed before use. Total suspended solids were determined by filtering a known water volume through the precombusted and preweighed Whatman 934-AH glass micorfibre filters immediately after returning from the field (APHA et al. 1989). The filters were then dried at 103 °C to a constant weight. Total suspended solids were calculated by subtracting the initial weight of the filter from the weight of the filter after drying at 103 °C. The filters were combusted again at 550 °C for one hour in a muffle furnace, cooled and reweighed. The organic fraction of the seston dry weight was that portion lost upon combustion. The inorganic fraction was determined from the residue remaining after combustion. All weighing was done on a Mettler Model H54AR balance.

The concentration of chlorophyll a corrected for phaeophytin was obtained using spectroscopy (APHA et al. 1989). Immediately after returning from the field Whatman GF/F glass microfibre filters were used for filtration and frozen over desiccant in the dark until analysis. During analysis filters were ground in acetone, centrifuged, and chlorophyll a

concentration was determined from the extract on a Gilford Response II spectrophotometer. To correct for phaeophytin the extract was treated with acid after chlorophyll a determination to convert all chlorophyll a to phaeophytin. The phaeophytin concentration was then determined and the corrected chlorophyll a concentration was determined through subtraction.

Nitrate nitrogen was determined with second-derivative spectroscopy following the method of Crumpton et al. (1992). Total nitrogen was also determined using second-derivative spectroscopy after potassium persulfate digestion of lake water samples, duplicate standards, and blanks in an autoclave at 18 psi for 40 minutes to convert all forms of nitrogen to nitrate (Crumpton et al. 1992). Urea standards consisting of 1 mg/L N were also digested to verify complete recovery. Standard concentration curves were produced on each day of sample analysis. The Gilford Response II spectrophotometer was connected to a computer which was used to calculate derivatives from absorbance. Nitrate standard solutions were prepared from a 1000 mg/L stock solution available commercially.

Ammonia nitrogen was measured with an ion-selective electrode (APHA et al. 1989) connected to an Orion Research Ionalyzer Model 250A. Ammonia standards and blanks were used to produce a concentration curve on each day of analysis. Ammonia standards were prepared from a 1000 mg/L stock solution available commercially. Organic nitrogen was calculated by subtracting nitrate nitrogen and ammonia nitrogen concentrations from the total nitrogen concentration.

Algae genera and biovolumes were determined by preserving 1 L of lake water sample with 10 ml of Lugols solution (APHA et al. 1989). The water sample was stored in a covered 1 L graduated cylinder until the phytoplankton completely settled. After settling the overlying water column was siphoned off and the algae decanted into a 50 ml screw-cap container. A Nikon compound microscope was calibrated with a stage micrometer and a whipple grid. During identification and enumeration a Palmer-Maloney counting cell was filled with sample and 10 fields were counted.

The membrane filtration method was used to determine fecal coliform bacteria concentrations at the inlets, tiles, and wildlife pen sites (Hach Company 1989). A known sample volume was passed through Gelman 0.45 µm membrane filters. The filters were then placed on an absorbent pad in a petri dish saturated with m-FC culture medium. The petri dish was then incubated in a water bath for 24 hours at 44.5 °C and fecal coliform colonies were counted. To simplify technique and minimize the possibility of contamination presterilized media, petri dishes with pads, dilution water, membrane filters, and disposable pipettes were purchased from Hach Company, Ames, Iowa. During analysis at least two 100 ml samples of dilution water were filtered and incubated to inspect for contamination.

Due to the great amount of algae in the lake, which would interfere with bacteria growth on a membrane filter, the most probable number method was used for lake samples (Hach Company 1989). The procedure used a presumptive coliform test using lauryl tryptose medium with a five-tube, three-dilution series. The tubes were incubated at 35 °C from a minimum of 24 hours to a maximum of 48 hours until gas formation determined the presence of coliform bacteria. Following the presumptive coliform test EC medium tubes were inoculated from the positive presumptive tubes and incubated at 44.5 °C for 24 hours. The presence of gas indicated a positive test for fecal coliform bacteria. The most probable number table was then used to determine fecal coliform concentrations from the number of positive tubes and dilutions. To simplify technique and minimize the possibility of contamination presterilized tubes, media, dilution water, and disposable pipettes were purchased from Hach Company, Ames, Iowa.

Statistical Analysis

Nonparametric statistical tests were used on bacteria data to show the effect of runoff events on bacteria concentrations, determine which sites were significantly higher than others, and to show relationships between concentrations of bacteria at inlets and beach sites.

Nonparametric tests were used because a normal distribution could not be assumed with the large variation in the data set.

To determine the influence of precipitation on bacterial concentrations at sites in the watershed, a sign test (Steele and Torrie 1980) was used to see if runoff events were transporting significantly higher concentrations of bacteria than routine sampling. The overall mean of routine samples was compared to means of runoff events. The number of runoff events above and below the overall routine sample mean was used in the test criterion.

To examine for differences in bacteria concentrations between sites in the watershed during both routine sampling and runoff events, a Friedman's multiple comparison test (Sprent 1989) was used to show which sites were significantly higher than others. Sites were ranked from lowest to highest by blocking over time and considering each site as a treatment. Differences in rank totals of each site were significant if they exceeded the least significant difference calculated from the test criterion.

Because the beach sites were a major concern due to health risks associated with fecal contamination a Spearman's coefficient of rank correlation (Steele and Torrie 1980) was used to determine if any relationship existed between inlets and beaches. Concentrations of bacteria at the inlets and beaches were ranked, and a correlation calculated from an analysis of the rankings. Tests were considered significant at $p<0.05$ for all procedures.

RESULTS AND DISCUSSION

Because levels of several water quality parameters, including fecal coliform bacteria, are affected by the amount of rainfall in a lake's watershed, comparisons were made of monthly rainfall totals between the 1980 diagnostic feasibility study, the post-restoration study years of 1987-1989, and 1993. Data were used from the National Oceanic and Atmospheric Administration weather station near Carroll. Comparison of rainfall data between 1980, 1987-1989, and 1993 are shown in Figure 5. The months June, July, and August are compared since all runoff events analyzed for fecal coliform were collected during these months in 1993. Total precipitation in 1993 during the month of June was lower than totals in 1980, 1988, and 1989, and only slightly higher than 1987. The July 1993 total was higher than all years compared. August totals show 1993 to be higher than 1980, 1988, and 1989, but lower than 1987. The variability between years should be considered when interpreting the results.

Fecal Coliform Bacteria

Fecal coliform results are presented for all sites in the Swan Lake watershed and lake sites L1, L2, L3, and the swimming beaches (Figures 6-8 and Tables 2-8). Mean values are presented for each site. Additional tables with original data are presented in Appendix C.

Comparison of inlets, tiles, and wildlife pen sites

Routine sampling Sites INCW, INWP1, and INN1 required rain induced runoff to be flowing. Wildlife pen number one (INWP1) was sampled twice during routine sampling and the northern inlet (INN1) on three occasions. These sites were likely flowing during routine sampling due to smaller rainfalls that did not meet the criterion for a runoff event, yet the routine sampling means for these sites may be more representative of runoff events. If the means for routine sampling are compared excluding these sites, then the tile draining part of the

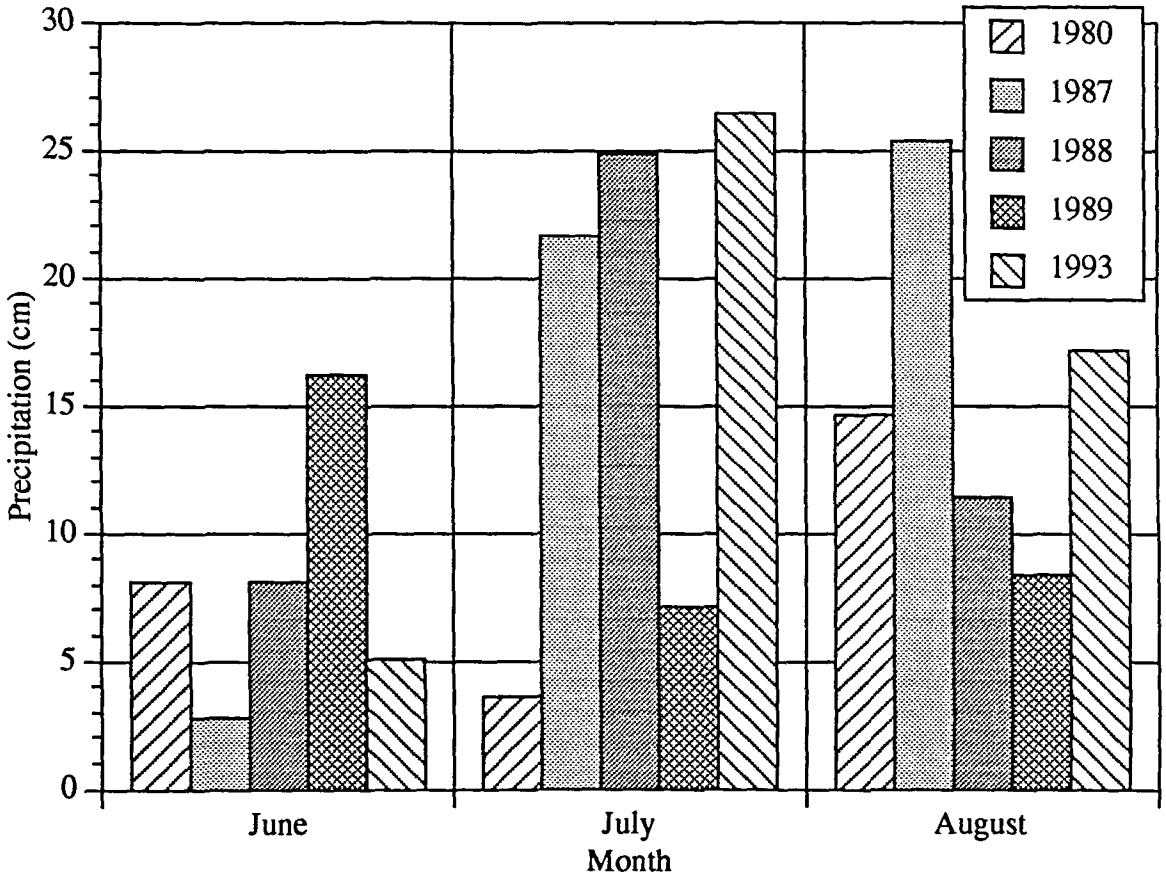

Figure 5. Monthly totals of precipitation near Carroll for June, July, and August of 1980, 1987-1989, and 1993

northern watershed (INN2) would have the highest mean of 552 colonies per 100 ml (Table 2). It is uncertain why this site had high concentrations of bacteria. The tile drains land in the park without any agricultural practices or wildlife pens. Possible sources of contamination may be from nearby restroom facilities or wildlife in the park. Wildlife pen number three (INWP3) had the next highest mean of 326 colonies per 100 ml, and the largest inlet (INW) had a mean of 251 colonies per 100 ml. The remaining site means were below or close to meeting the water quality standard (200 fecal coliform colonies per 100 ml) during routine sampling.

Table 2. Routine sample collection fecal coliform means (colonies/100 ml) and standard errors for Swan Lake inlets, tiles, and wildlife pen sites in 1993

Site	Sample Size	Mean	Standard Error
INW	9	251	55
INCE	9	212	128
INCW	0		
INSW	9	170	42
INWP3	8	326	79
INS	9	13	5
INSE	8	164	40
INWP1	2	962	818
INN1	3	961	493
INN2	7	552	249

Runoff events vs. routine sampling Data were collected on six runoff events that met the criterion of at least 2.5 cm (1 inch) of rain in a 24-hour period. Twenty four-hour rainfalls ranged from 2.5 cm to 18 cm (7 inch) in this study; the unusual 18 cm rainfall occurred on July 9, 1993. On two occasions the criterion was met on consecutive days.

The mean for each site during routine sampling (Table 2) was usually lower than the concentrations on runoff events (Figure 6). Only one runoff event, on June 14, 1993, was lower than some of the site means for routine sampling. The runoff event on August 30, 1993 clearly had a great impact on water quality at the tile site on the southwest portion of the watershed (INCE) and the main west inlet (INW). This event followed two consecutive days of rain. Even though there were heavier rain events during the summer, bacterial counts never approached these levels. Wildlife pen number one (INWP1) had consistently high bacterial counts during runoff events. This site was flowing only twice during routine sampling. The other sites did not have high concentrations like those measured at sites INCE, INW, and INWP1, but most runoff events had a higher impact on water quality at each site than non-runoff conditions.

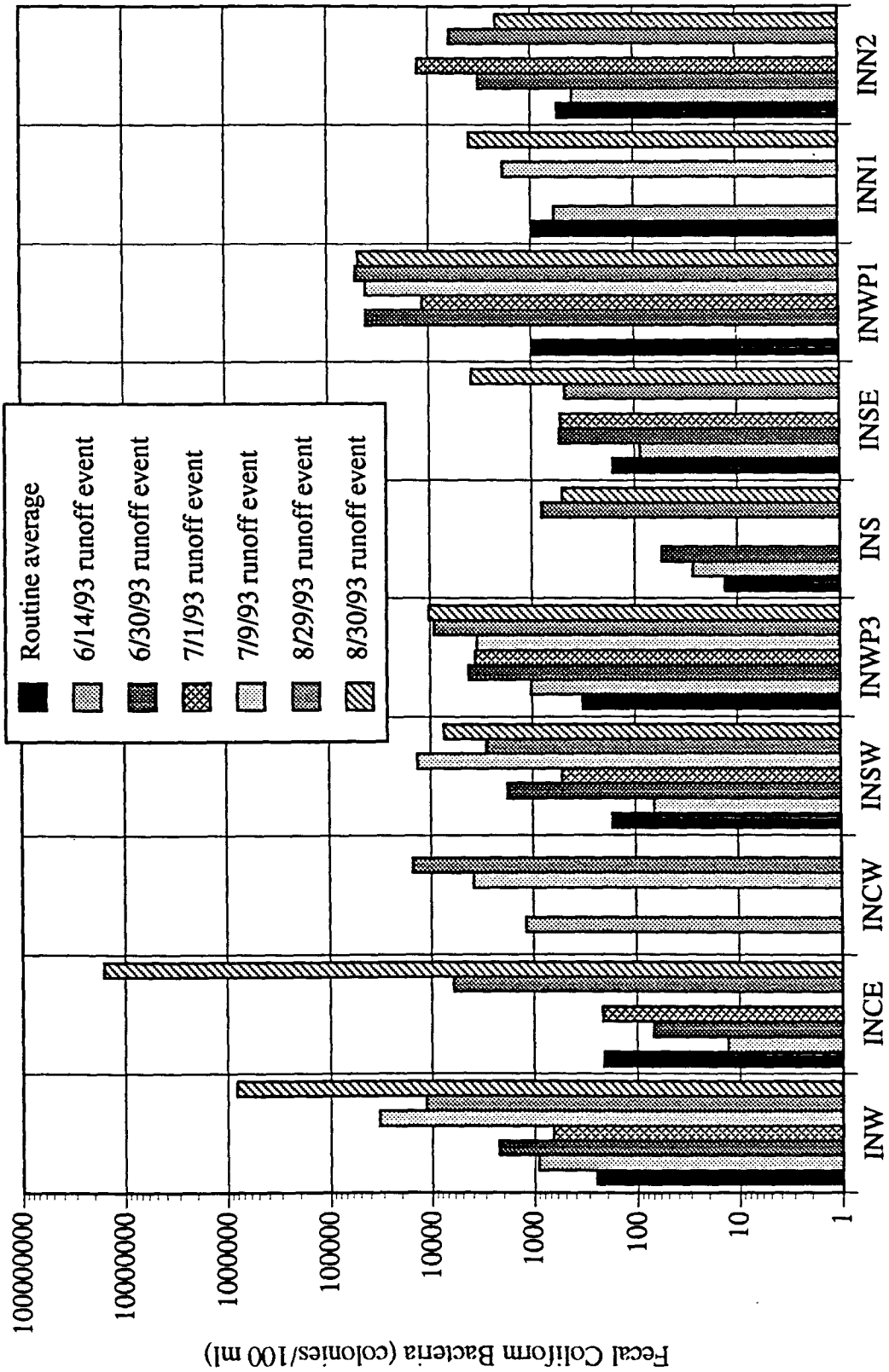

Figure 6. Routine sample collection means shown with runoff concentrations (log scale) of fecal coliform bacteria in 1993 for Swan Lake inlets, tiles, and wildlife pen sites

To determine the affect rainfall-induced runoff had on bacterial concentrations, a sign test (Steel and Torrie 1980) was used to compare the mean of routine sampling dates and means of runoff events (Table 3). The sign test showed inlets, tiles, and wildlife pen runoff to have significantly higher concentrations ($p<0.05$) of fecal coliform bacteria during runoff events. Several researchers have observed significant increases in indicator bacteria concentrations during runoff events in agricultural watersheds (Robbins et al. 1972; Doran and Linn 1979; Dudley and Karr 1979; and Patni et al. 1985). Runoff events may act as a flushing mechanism after the buildup of bacteria in watersheds between rainfalls.

Table 3. Means of fecal coliform bacteria (colonies/100 ml) and standard errors for all sampling dates including routine sampling and runoff events. Means were calculated for each date from all flowing inlets, tiles, and wildlife pen sites in the Swan Lake watershed in 1993

	Sample Size	Mean	Standard Error
Routine Sample Dates			
6/2/93	8	307	210
6/9/93	6	56	28
6/23/93	5	205	117
7/14/93	8	502	117
7/28/93	9	614	240
8/9/93	7	305	125
9/14/93	7	245	80
9/26/93	7	101	30
10/12/93	7	68	40
Runoff Sample Dates			
6/14/93	9	475	155
6/30/93	8	6,632	4,900
7/1/93	7	4,271	2,100
7/9/93	6	16,112	6,900
8/29/93	9	11,548	5,300
8/30/93	9	1,876,292	1,770,000

To examine for differences between sites in the watershed during runoff events a Friedman's test (Steel and Torrie 1980) was used to verify overall significance. Sites were ranked from lowest to highest on each runoff sampling date, and the sum of the ranks from each site were used in the test criterion. The test indicated there were significant differences ($p<0.05$) between sites during runoff events. The Friedman's test was then extended to perform multiple comparisons (Sprent 1989) between the rank totals of individual sites. The magnitude between differences of rank totals is significant ($p<0.05$) if they exceed the least significant difference calculated from the rank sum of squares. Table 4 lists the difference between rank totals and indicates which differences were significant. Wildlife pen number one (INWP1), the main west inlet (INW), and wildlife pen number three (INWP3) had significantly higher concentrations of bacteria during runoff events than INS, INN1, INSE, and INCE. Site INN2 was significantly higher than site INS.

Table 4. Differences between rank totals of Swan Lake inlets, tiles, and wildlife pen sites for fecal coliform bacteria during runoff events in 1993

Site	INWP1	INW	INWP3	INN2	INSW	INCW	INCE	INSE	INN1
INS	32.5*	32.5*	30.5*	20.0*	17.5	15.0	12.0	9.0	6.0
INN1	26.5*	26.5*	24.5*	14.0	11.5	9.0	6.0	3.0	
INSE	23.5*	23.5*	21.5*	11.0	8.5	6.0	3.0		
INCE	20.5*	20.5*	18.5*	8.0	5.5	3.0			
INCW	17.5	17.5	15.5	5.0	2.5				
INSW	15.0	15.0	13.0	2.5					
INN2	12.5	12.5	10.5						
INWP3	2.0	2.0							
INW	0.0								

* designates $p<0.05$.

Combined routine samples and runoff samples Because bacterial contamination is greatly influenced by precipitation, the impact of individual sites in the Swan Lake watershed should be assessed including both routine sampling and runoff events to examine inputs over the entire study period. Mean values of fecal coliform bacteria, calculated from all samples including both routine sampling and runoff events, for inlets, tiles, and wildlife pens are shown in Figure 7. The great amount of variation between routine sampling and runoff events is the reason for the high standard errors.

The tile draining the cornfield on the southwest part of the watershed (INCE) had the highest mean of 1,140,000 colonies per 100 ml of sample (Figure 7). This site was sampled on all occasions except one time in July when the area received 18 cm of rain and the tile was below water. The reason for this site's high mean is due to a runoff event on August 30, 1993 when the concentration of bacteria was 16 million colonies per 100 ml of sample. This concentration was by far the highest measured during the study period. Although this site generally had relatively low concentrations throughout the study when compared to other sites, this high concentration is of great concern. Baxter-Potter and Gilliland (1988) identified concentrations of fecal coliform bacteria ranging from 1.3 million to 79 million colonies per 100 ml in cattle feedlot runoff. The high concentration of bacteria suggests animal waste from the cattle feedlot, bordering the cornfield that the tile drains, may have entered the tile system during this runoff event.

The second highest mean of 57,300 colonies per 100 ml of sample (Figure 7) is from the main west inlet (INW). This site is located on the lake's largest inlet and was flowing during all routine sample collections and runoff events. This inlet receives the drainage from site INCE and also had a high mean due to the runoff event on August 30. Even though this event increased the mean of this site, it often had some of the highest fecal coliform counts during the study period.

Wildlife pen number one (INWP1) had the third highest mean of 28,000 colonies per

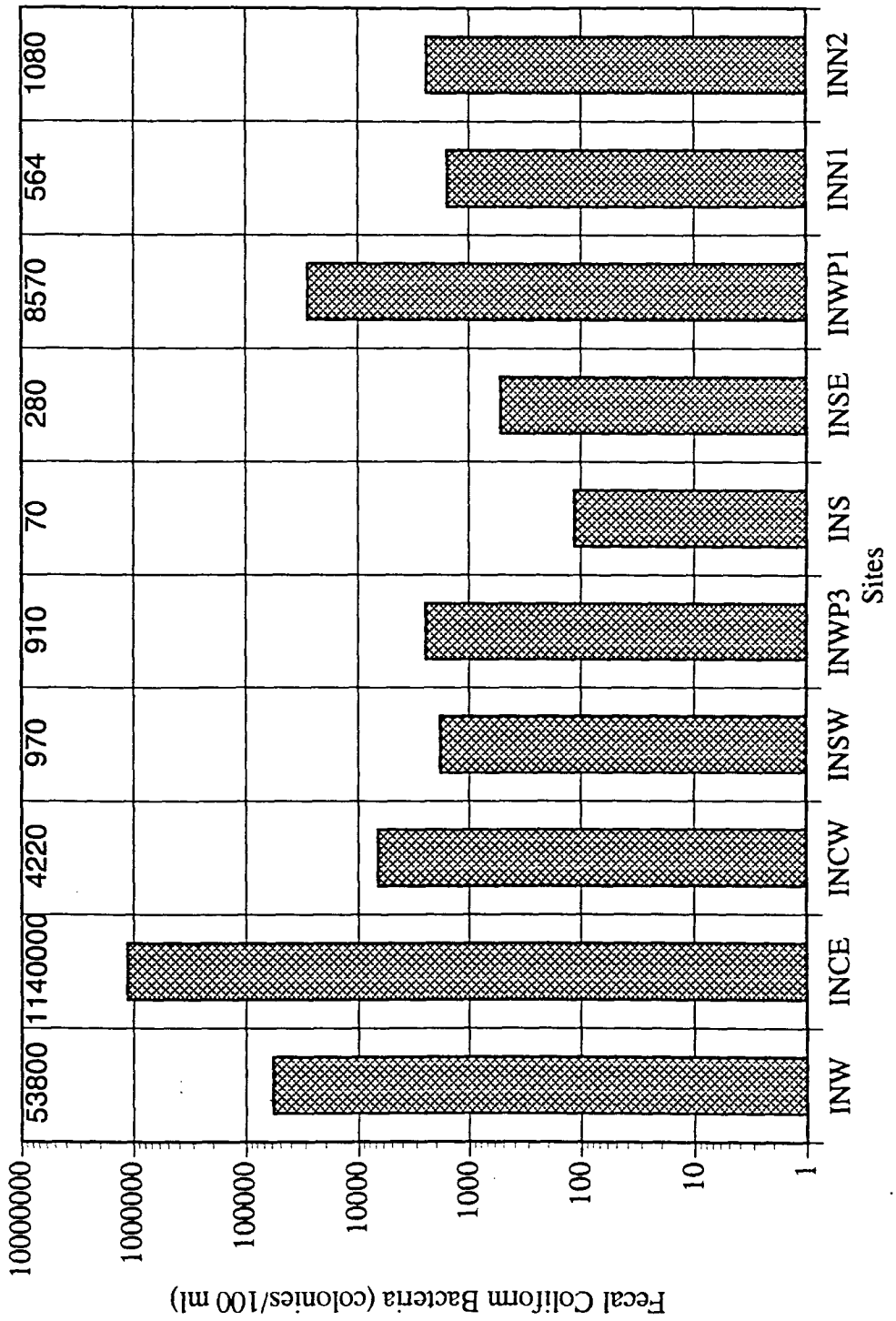

Figure 7. Mean bacteria concentrations (log scale) in 1993 for all samples including both routine sampling and runoff events at Swan Lake inlets, tiles and wildlife pen sites in 1993. Standard errors are listed across the top row for each site

100 ml of sample (Figure 7). When this site was flowing bacterial counts were quite high. This pen housed whitetail deer, wild turkeys, and various waterfowl species. Animal waste accumulating in the pen prior to rainfalls would likely contain high concentrations of bacteria that could become available to runoff during rainfall events. Because this site only flowed following rainfalls high concentrations of bacteria were likely flushed out of the pen area in a brief period of time.

The next highest mean is from site INCW (Figure 7). Site INCW, located on the main west inlet above the tile site INCE, was flowing only three times during the study, each of these during a runoff event. Although this site was not flowing very often, when it was it contained high concentrations of bacteria. This site had the fourth highest mean of 6,650 colonies per 100 ml of sample.

Wildlife pen number three (INWP3), the northern inlet (INN1), the inlet entering the lake from the southwest (INSW), and the northern tile (INN2) had similar means (Figure 7). Site INWP3 was flowing on all but one sample collection and had a mean of 2,460 colonies per 100 ml of sample. It had a lower mean than INCE, INW, INWP1, and INCW because it did not have an extremely high concentration during any one event. However, its concentrations were often among the highest throughout the study period when compared to other sites. Because this inlet was flowing on all but one occasion it may have been supplying the lake with a continuous input of bacteria. Site INN2 had the next highest mean of 2,390 colonies per 100 ml of sample. This site was flowing during most of the study period and sometimes had high bacteria counts. Site INSW, located on the inlet entering the lake from the southwest below the lagoons which received runoff from the cattle feedlot, had a mean of 1,840 colonies per 100 ml of sample. When compared to the means of other sites this suggests that the lagoons were functioning in retaining most of the runoff. There was some concern that the lagoons were overflowing during runoff events. If individual runoff events are compared with the wildlife pens, site INSW only exceeded wildlife pen number three (INWP3) on July

9, 1993 when the area received 18 cm of rain, and only exceeded wildlife pen number one (INWP1) on June 14, 1993 when that site was not flowing (Figure 6). The northern inlet (INN1) was not always flowing and when it was it usually had lower bacterial counts than the other sites. This site had a mean of 1,550 colonies per 100 ml of sample (Figure 7).

The inlet entering the lake from the southeast part of the watershed (INSE) and the tile discharging into the lake from the south (INS) were flowing throughout most of the study period and had the lowest concentrations of bacteria. Sites INSE and INS had means of 520 and 115 colonies per 100 ml of sample, respectively (Figure 7).

Overall contributions of bacteria Because some sites were not flowing on each sample collection, their overall contribution of bacteria to the lake may not have been as large as those sites that flowed throughout the study period. To examine the contributions from all sites over the entire study period, those sites that were not flowing on a sampling date were deemed as contributing no bacteria to the lake on that particular date. Sites that were not flowing on a sampling date received the lowest ranks in a Friedman's multiple comparison test (Sprent 1989). When examined in this way, sites INWP3 and INW were significantly higher than six of nine sites (Table 5). These two sites were flowing throughout the study period and they may have been regularly supplying the lake with bacteria.

Frequency of sites having high or low concentrations of bacteria The great variation between sites and between routine sampling and runoff events adds to the difficulty of determining the major sources of fecal coliform bacteria. Even though most sites can be high at times, there are some that are more likely to be higher than others. The probability that a site will be among the lowest, intermediate, or the highest of all sites is listed in Table 6. The probabilities are for any given date and include those times that sites were not flowing. The probabilities were calculated by ranking each site from lowest to highest on each sampling date. The lowest site on a particular date would receive a rank of 1 and the highest a rank of 10. Ties occurred where more than one inlet was not flowing and each of these sites received the mean

Table 5. Differences between rank totals of Swan Lake inlets, tiles, and wildlife pen sites for fecal coliform bacteria including routine samples and runoff events in 1993

Site	INWP3	INW	INSW	INN2	INCE	INWP1	INSE	INN1	INS
INCW	70.0*	66.0*	50.0*	46.0*	35.5*	29.0*	28.5	11.5	4.5
INS	65.5*	61.5*	45.5*	41.5*	31.0*	24.5	24.0	7.0	
INN1	58.5*	54.5*	38.5*	34.5*	24.0	17.5	17.0		
INSE	41.5*	37.5*	21.5	17.5	7.0	0.5			
INWP1	41.0*	37.0*	21.0	17.0	6.5				
INCE	34.5*	30.5*	14.5	10.5					
INN2	24.0	20.0	4.0						
INSW	20.0	16.0							
INW	4.0								

* designates $p<0.05$.

Table 6. Probabilities of Swan Lake inlets, tiles, and wildlife pens on any given date to be among the lowest, intermediate, or highest of all sites in 1993

	Probabilities		
Site	Group 1	Group 2	Group 3
INW	.00	.46	.53
INCE	.40	.33	.26
INCW	.80	.06	.13
INSW	.13	.60	.26
INWP3	.06	.20	.73
INS	.86	.13	.00
INSE	.33	.53	.13
INWP1	.60	.00	.40
INN1	.60	.26	.13
INN2	.26	.33	.40

Group 1= four lowest sites.
Group 2= next three lowest sites.
Group 3= three highest sites.

rank. The largest number of inlets not flowing on a particular date was four. Each of these inlets received a rank of 2.5, and were among the lowest four for that particular date. The probability for each site occurring in group 1 (ranks 1-4), group 2 (ranks 5-7), and group 3 (ranks 8-10) was calculated from a total of fifteen sampling dates.

Wildlife pen number three (INWP3) had the highest probability on any given date to be among the highest three sites (Table 6), and this site was flowing on all but one occasion. The main west inlet (INW) had the second highest probability of being among the highest three sites and was never among the lowest four. The tile that discharged into the lake from the south (INS) had the highest probability of being among the lowest four sites and was never among the highest three. The probabilities for wildlife pen number one (INWP1) show this site to be among the highest or lowest depending upon whether it was flowing. The probabilities for the tile site INCE indicate this site would more likely be among the lowest than the highest, even though it contained the highest bacterial count of the study from one runoff event.

Comparison of Swan Lake inlets with control inlet To observe background levels of bacteria and determine if Swan Lake inlets were carrying unusually high concentrations of bacteria, a comparison was made with an inlet outside of the Swan Lake watershed. Originally it was proposed that a nearby watershed, with a tributary not affected by animal waste from the production of livestock, be selected for the comparison. However, a tributary could not be found in Carroll County that was not potentially affected by animal waste. Therefore, a site was selected on the Big Creek Inlet in Boone County, Iowa. The Big Creek site drained a much larger watershed and had a higher flow rate than a typical Swan Lake inlet, but a comparison with this inlet is meaningful because the Big Creek site was not affected by animal waste. There were cornfields upstream from the Big Creek sampling site, but no cattle grazing or spreading of animal waste occurred. Due to differences in rainfall between the two watersheds only the routine sample collections were compared. The fecal coliform mean for

the Big Creek Inlet was 240 colonies per 100 ml. If compared with Swan Lake sites (Table 2), excluding wildlife pen number one (INWP1) and the northern inlet (INN1) which required rain induced runoff to be flowing, there were 4 sites below and 3 sites above the Big Creek mean. Sites above the Big Creek mean, INW, INWP3, and INN2, may have been transporting higher concentrations of bacteria than would be expected from background levels from areas not subjected to livestock waste.

Lake sites

Previous water quality monitoring at Swan Lake measured concentrations of fecal coliform bacteria at the east swimming beach. To gain perspective on the concentrations of bacteria measured in the lake in 1993, a comparison of 1980, 1987-1989, and 1993 means are shown in Figure 8. The 1980 mean was well below the water quality standard, however, more recent years have been above the standard. The highest mean was in 1987; 1988 and 1989 means were just below or close to meeting the standard. The 1993 mean was further above the water quality standard than 1988 and 1989 means.

The water quality monitoring program that was conducted in 1987 through 1989 (IDNR 1993) measured relatively high concentrations of fecal coliform bacteria at the east swimming beach during the summer months. These concentrations exceeded the water quality standard for protection of primary contact uses in 8 of 22 samples. This study found the east and west beaches also exceeded the water quality standard (Table 7). The east beach exceeded the standard in 4 of 9 samples and the west beach in 3 of 9 samples. Lake site L1 never exceeded the standard. However, lake site L2 was above the standard on 4 of 9 samples, and lake site L3 on 2 of 8 samples.

All lake sites were sampled for fecal coliform bacteria approximately 0.5 m below the surface. Lake site L1 was located at the deepest portion of the lake and never exceeded the water quality standard. This may have been due to dilution in the deeper area of the lake. Bacteria that entered the lake may also have settled to the bottom sediments before reaching site

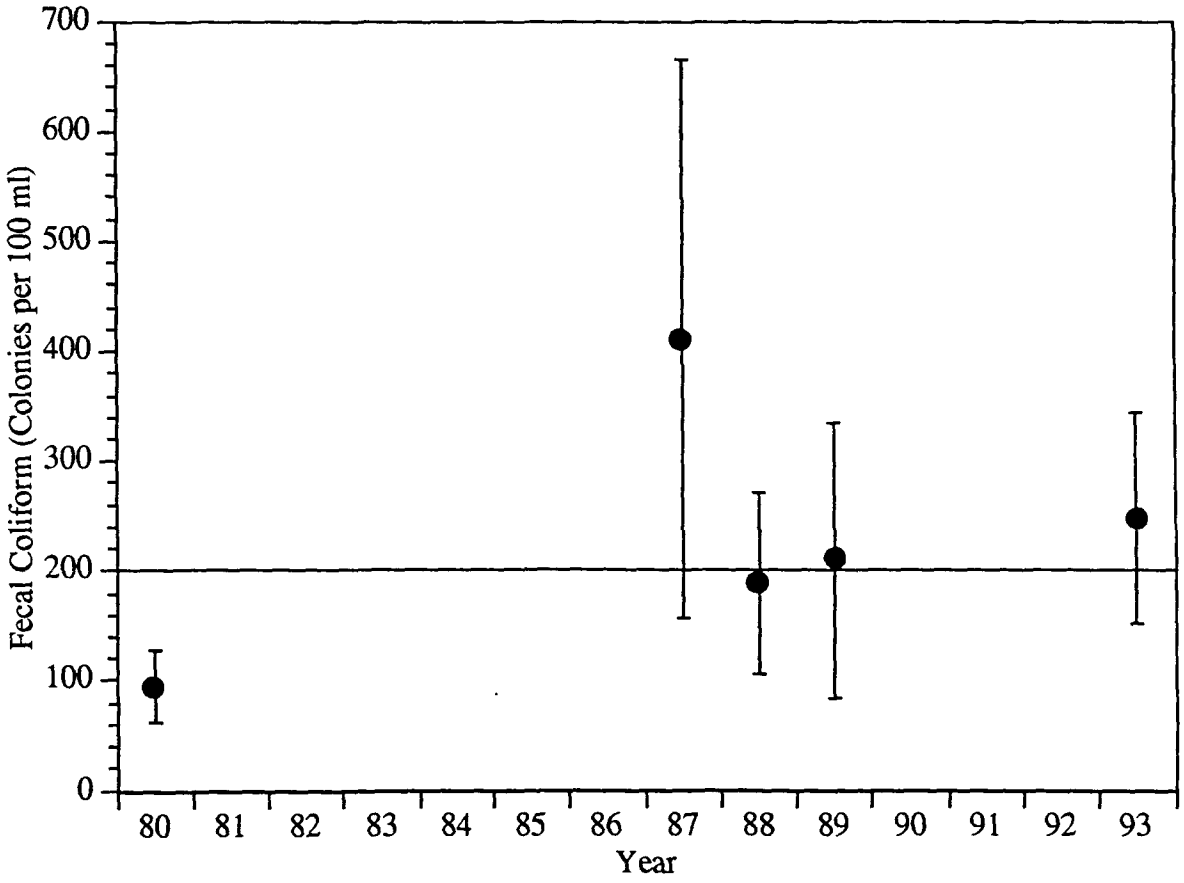

Figure 8. Seasonal means of fecal coliform bacteria at Swan Lake east swimming beach with +/- one standard error. The water quality standard is indicated by a line at 200 fecal coliform colonies per 100 ml. Means were calculated for 1980 and 1987-1989 from data collected by Bachmann et al. (1982) and IDNR (1993)

L1 at the lower end of the lake near the dam. Gannon et al. (1983) demonstrated sedimentation was important in fecal coliform disappearance in the lower end of Lake Ford, Michigan. Bergstein and Stone (1991) also found sedimentation to influence fecal coliform distribution in Lake Kinneret, Israel. They found bacterial densities in surface water reduced with distance from the main inlet, while bacterial densities in the sediment and bottom waters

Table 7. Fecal coliform bacteria counts per 100 ml in 1993 at various sites in Swan Lake

Date	Site				
	East Beach	West Beach	L1	L2	L3
6/2/93	58	34	78	56	20
6/29/93	290	300	14	220	17
7/14/93	500	900	155	300	140
7/28/93	110	195	140	280	220
8/9/93	220	170	100	140	
8/30/93[a]	900	5000	90	800	3000
9/14/93	110	170	40	130	140
9/26/93	40	20	20	2	20
10/12/93	2	2	0	2	2

[a] runoff event.

increased at the farthest sampling station from the inlet. Sedimentation has been shown to play a role in the distribution of bacteria in lakes and is likely affecting distribution in Swan Lake.

Lake sites L2 (mid-lake) and L5 (east beach) were above the water quality standard more often than the other lake sites. These sites were likely affected most by inputs of bacteria from the watershed because the sites were located near mid-lake with inlets entering the lake from the west, north, and south. Lake sites L3 (upper end of lake) and L4 (west beach) did not exceed the water quality standard as often as sites L2 and L5. These sites, located at the upper end of the lake, were likely affected by the main west inlet (INW) and the inlet entering the lake from the southwest (INSW).

The highest bacterial counts measured in the lake were on August 30, 1993 during the runoff event where the concentration of fecal coliform bacteria at tile site INCE was 16 million colonies per 100 ml. The east and west swimming beaches exceeded the water quality standard by 4.5 and 25 times, respectively. On July 14, 1993 the east and west beaches also had high bacterial counts. This may be related to the July 9, 1993 rainfall when the area received 18 cm

of rain. A rainfall of this magnitude likely increased the flow at all sites for several days, increasing bacterial concentrations in the lake.

The concentrations of bacteria at the east and west beaches are the greatest concern because of public health hazards associated with fecal pollution. The inlets that had the greatest affect on bacteria concentrations at the east and west beaches were determined by using Spearman's coefficient of rank correlation (Steel and Torrie 1980). To identify problem areas in the watershed, inlets in close proximity to each other were grouped together in pairs and their concentrations of bacteria were combined. Inlets that were paired included those entering the lake from the west (INW and INSW), south (INWP3 and INS), southeast (INSE and INWP1), and north (INN1 and INN2). Bacterial counts from the beaches and each pair of inlets were ranked from lowest to highest on each routine sampling date. The correlation was calculated from an analysis of the rankings to determine if any relationships between the beaches and inlets existed. The correlations should be interpreted with care because they only represent routine sample collections.

Table 8 shows the east and west beaches were most highly correlated with inlets entering the lake from the west (INW and INSW) and north (INN1 and INN2). The inlets entering the lake from the north are located at approximately mid-lake between the two beaches and may be affecting bacteria levels at both beaches. The main west inlet (INW) and the inlet entering the lake from the southwest (INSW) are the largest inlets flowing into the lake, and this may account for the strong influence these inlets may have had on the beaches during routine sampling.

Conclusion on fecal coliform bacteria

Most sites in the Swan Lake watershed were below or close to meeting the water quality standard during routine sampling. Sites INN2 and INWP3 had routine sample means exceeding the water quality standard by approximately 1.5 and 2.5 times, respectively. These sites were also above the Big Creek control inlet mean of 240 colonies per 100 ml during

Table 8. Correlations between beach sites and paired sites in the Swan Lake watershed for routine sample collections of fecal coliform bacteria in 1993

Beach	Paired Sites	Spearman Correlation Coefficient r_S
East	INW, INSW	0.88*
	INN1, INN2	0.83*
	INWP3, INS	0.62
	INWP1, INSE	0.47
West	INW, INSW	0.93*
	INN1, INN2	0.90*
	INWP3, INS	0.74
	INWP1, INSE	0.52

* designates $p<0.05$.

routine sampling. The source of contamination at tile site INN2 is unknown. Possible sources include effluent from nearby restroom facilities entering the tile line or wildlife in the park. Wildlife pen number three houses buffalo and runoff from this area likely transports animal waste from the pen.

Runoff events were shown to have significantly higher concentrations of bacteria than routine sampling. Sites that were shown to be the highest during runoff events included INWP1, INW, and INWP3. These sites were significantly higher than four of nine sites in the watershed. The runoff event that occurred on August 30, 1993 caused tile site INCE to have extremely high bacterial counts. This happened only one time during the study period, but created very poor water quality. Animal waste from the cattle operation likely entered the tile system on this particular date. The lake was also sampled on August 30 and all lake sites, except L1, exceeded the water quality standard. Bacterial counts in the lake on August 30 were the highest measured during the study period.

The high bacterial counts on August 30 were measured after two consecutive days of rain which created heavy runoff conditions. However, there was one runoff event on July 9 that was larger than the August 30 runoff, and some of the sites that were sampled on July 9 had lower bacterial counts than on August 30. This may have been due to the time of sample collection during the July 9 runoff event. Sampling was conducted after 18 cm of rain had fallen. Most of the bacteria may have been flushed from the watershed after the start of this intense storm event. Doran and Linn (1979) collected samples three times during each runoff event and found bacterial counts were usually highest during peak runoff flows and decreased with time thereafter. Following the 18 cm rainfall, sampling may not have taken place until after higher concentrations of bacteria were already transported from the watershed. In addition, due to the 18 cm of rain that fell on July 9 the tile sites INCE, INS, and INN2 were below water and could not be sampled. The bacterial counts at these sites during the July 9 runoff event are unknown. The July 9 runoff event may have had high bacterial counts like the August 30 runoff event, but due to the time of sampling and the number of sites that could not be sampled higher bacterial counts were not detected.

To examine inputs over the entire study period means were calculated from all samples including routine sampling and runoff events. Those sites that had the highest means included INCE, INW, and INWP1. Sites INCE and INW had high means as a result of the August 30 runoff event. Site INWP1 had a high mean due to consistently high concentrations of bacteria during runoff events. Because all inlets were not flowing all of the time overall contributions of bacteria to the lake were also examined including those times that sites were not flowing. Sites INWP3 and INW were significantly higher than six of nine sites. These two sites also had the highest probability of being among the highest three sites on all sampling dates. These sites were flowing throughout the study period and may have been consistently adding large numbers of bacteria to the lake.

Nutrients and Other Limnological Parameters

Concentrations of phosphorus, nitrogen and suspended solids from Swan Lake inlets, tiles, and wildlife pens were determined and results are presented here. In addition, results are presented for lake sites L1, L2, and L3 which were monitored for phosphorus, nitrogen, suspended solids, chlorophyll a, transparency, temperature, dissolved oxygen, pH, and total alkalinity. Comparisons are also made with previous water quality studies conducted on Swan Lake. Only lake site L1 at 0.5 m was used for comparison, because this was the only site sampled in earlier studies. Previous studies include the diagnostic feasibility study conducted in 1980 by Bachmann et al. (1982), the post-restoration study from 1987-1989 (IDNR 1993), and data collected in 1990 for the most recent lake classification study conducted by Bachmann et al. (1994). Where comparisons are made means are used from the May-September periods in 1980, 1987-1989, and 1993. However, data from 1990 were only collected one time during the months of June, July, and August. Even though sample sizes are smaller for 1990, the means likely represent yearly trends in water quality. Additional tables with original data are presented in Appendix C.

Phosphorus

Inlets, tiles, and wildlife pens Phosphorus is a major nutrient affecting algal biomass. The 1987-1989 post-restoration study detected very high levels of total phosphorus in Swan Lake (IDNR 1993). It was suggested that high levels of phosphorus may have resulted from nutrient management problems in the watershed. Concern has been expressed about application of animal waste on the watershed during winter months and effects of runoff on Swan Lake water quality. Studies have shown increased nutrient loss from watersheds when animal manure was applied during winter months (Klausner et al. 1976 and Young and Mutchler 1976). Application of animal waste on fields in the Swan Lake watershed during winter months may have contributed high levels of phosphorus to runoff derived from snow melt and rainfall.

This study found high phosphorus concentrations during runoff events at most sites in the Swan Lake watershed (Table 9). Site INCW, located on the main west inlet above tile site INCE, had the highest mean of 0.773 mg/L of total phosphorus during runoff events. Wildlife pen number one (INWP1) had the second highest mean of 0.563 mg/L. Other sites with high means during runoff events included the main west inlet (INW), the tile that discharges into the main west inlet (INCE), the inlet entering the lake from the southwest (INSW), and wildlife pen number three (INWP3).

Total phosphorus concentrations in routine samples were relatively low when compared to runoff events (Table 9). Because sites INCW, INWP1, and INN1 required rain induced runoff to be flowing, the routine means for these sites may be more representative of runoff events. Site INCW was never flowing during routine sampling, but sites INWP1 and INN1 were flowing 2 and 3 times respectively during routine sampling. These sites were likely flowing during routine sample collection because of smaller rainfalls prior to sampling that did not meet the criterion for a runoff event. If these sites are excluded when comparing routine sample means, then site INSW had the highest mean of 0.120 mg/L of total phosphorus. All other sites had a mean less than 0.100 mg/L of total phosphorus during routine sampling.

To examine total phosphorus inputs over the entire study period means were also calculated for all samples including both routine sampling and runoff events (Table 9). The total phosphorus means, including both routine sampling and runoff events during the 1993 study period, indicated INCW and INWP1 had the highest means. These sites had the highest means because they required rain induced runoff to be flowing which contributed larger inputs of total phosphorus than non-runoff conditions. The three highest means from sites that were flowing throughout the study period were from sites INW, INSW, and INCE. Site INW had a mean of 0.281 mg/L and site INSW had a similar mean of 0.275 mg/L. Site INCE was slightly lower with a mean of 0.234 mg/L. The remaining sites were near or below 0.200 mg/L of total phosphorus.

Table 9. Mean total phosphorus concentrations (mg/L) for Swan Lake inlets, tiles, and wildlife pen sites in 1993 for routine sampling, runoff events, and all samples including both routine sampling and runoff events

Samples taken during runoff events

Site	Sample Size	Mean	Standard Error
INW	12	0.492	0.161
INCE	9	0.476	0.234
INCW	8	0.773	0.193
INSW	12	0.430	0.131
INWP3	13	0.345	0.138
INS	12	0.223	0.110
INSE	12	0.244	0.088
INWP1	9	0.563	0.102
INN1	9	0.296	0.099
INN2	11	0.177	0.055

Samples taken during routine sampling

Site	Sample Size	Mean	Standard Error
INW	12	0.070	0.007
INCE	12	0.052	0.004
INCW	0	0.000	0.000
INSW	12	0.120	0.007
INWP3	12	0.047	0.004
INS	12	0.031	0.002
INSE	12	0.047	0.004
INWP1	2	0.712	0.414
INN1	3	0.299	0.177
INN2	10	0.088	0.006

Samples taken during both routine sampling and runoff events

Site	Sample Size	Mean	Standard Error
INW	24	0.281	0.090
INCE	21	0.234	0.107
INCW	8	0.773	0.193
INSW	24	0.275	0.072
INWP3	25	0.202	0.077
INS	24	0.127	0.057
INSE	24	0.146	0.048
INWP1	11	0.590	0.101
INN1	12	0.296	0.082
INN2	21	0.135	0.030

Lake sites Swan Lake had very high levels of total phosphorus in 1987-1989 when compared to 1980 levels, and also when compared to other lakes in Iowa sampled during this same time period (IDNR 1993). After lake deepening during restoration it was suggested that total phosphorus concentrations might decline due to dilution and reduced recycling of phosphorus from the sediments (Bachmann et al. 1982). However, the means were much higher in 1987, 1988, and 1989 following restoration, with total phosphorus means of 0.613, 0.570, and 0.610 mg/L, respectively (IDNR 1993).

The total phosphorus mean in 1993 was compared with previous water quality studies conducted on Swan Lake (Figure 9). The 1993 mean of 0.186 mg/L of total phosphorus was much lower than the post-restoration study means. The highest level of total phosphorus measured in Swan Lake during the 1993 study period was 0.438 mg/L. This concentration was measured five days after an 18 cm rainfall on July 9, 1993. The unusually high amount of precipitation probably increased the amount of phosphorus entering the lake in runoff and would account for this high concentration. Even though this rare event may have increased levels in the lake, 1993 levels were not nearly as high as those measured in the post-restoration study.

Lake sites L1, L2 and L3 had similar means of total and soluble reactive phosphorus (Table 10). The total phosphorus mean at lake site L1 was 0.186 mg/L, and 0.180 mg/L over all depths. Site L2 had a mean of 0.178 mg/L of total phosphorus and site L3 had 0.184 mg/L. The similar means of L1 (deepest portion of the lake), L2 (mid-lake), and L3 (upper end of the lake) indicated phosphorus levels were evenly distributed throughout the lake in 1993.

It is unknown why total phosphorus concentrations have apparently decreased in the lake. Differences in total phosphorus levels between the post-restoration study and this 1993 study may have resulted from different laboratories using different analytical methods. Even though it seems phosphorus levels have decreased in the lake a comparison of 1980, 1990, and 1993 total phosphorus levels, which were analyzed by the ISU Limnology Lab using the same

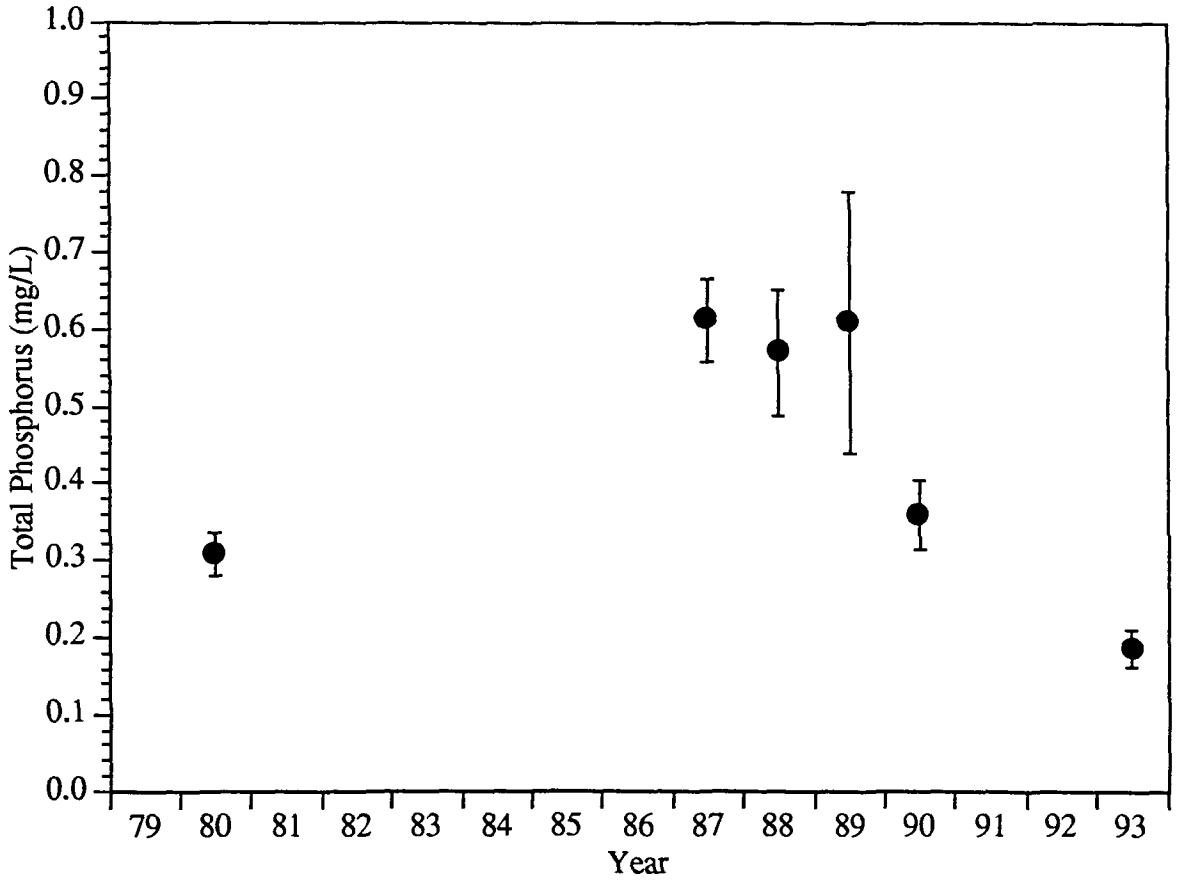

Figure 9. Seasonal means of total phosphorus at Swan Lake with +/- one standard error. Means are from the May-September period in 1980, 1987-1989, and 1993. The 1990 mean is from the June-August period. Means were calculated for 1980, 1987-1989, and 1990 from data collected by Bachmann et al. (1982), IDNR (1993), and Bachmann et al. (1994)

analytical method, shows lower means than the post-restoration study. The 1980, 1990, and 1993 studies indicated that algal blooms are still a water quality problem. Bachmann and Jones (1974) demonstrated from a diverse group of lakes that phosphorus inputs would have to be reduced below about 0.02 mg/L before noticeable improvement in water transparency would be achieved as a result of reduced algal crops. Marsden (1989) showed lakes with mean total

Table 10. Means of total phosphorus and soluble reactive phosphorus at various sites in Swan Lake in 1993

Site	Depth (m)	Total Phosphorus (mg/L)			Soluble Reactive Phosphorus (mg/L)		
		Sample Size	Mean	Standard Error	Sample Size	Mean	Standard Error
L1	0.5	13	0.186	0.024	13	0.011	0.007
L1	2.0	13	0.179	0.024	13	0.010	0.007
L1	3.5	13	0.174	0.021	13	0.012	0.008
L1	All	39	0.180	0.013	39	0.011	0.004
L2	0.5	13	0.178	0.024	13	0.010	0.007
L3	0.5	13	0.184	0.025	13	0.010	0.007

phosphorus concentrations greater than 0.100 mg/L had little reduction in phytoplankton biomass, unless the reduction in phosphorus loading was greater than 60 percent. This indicates Swan Lake is still receiving enough phosphorus to support algal blooms regardless of the apparent decrease in phosphorus since the post-restoration study.

Another source of phosphorus for phytoplankton may be internal loading from resuspension of nutrient rich sediment in the shallow portions of the lake. Wind-driven resuspension of sediments could induce regeneration of phosphorus from the sediments. Carper and Bachmann (1984) identified wind resuspension in a shallow lake in north-central Iowa. They found inorganic suspended solids increased in the lake on days when wind velocity exceeded a calculated critical velocity. In addition to wind-driven resuspension, carp (*Cyprinus carpio*) have reestablished populations in the lake since restoration. Carp foraging in bottom sediments may also contribute to resuspension of nutrient rich sediment. In highly eutrophic lakes release of phosphorus from sediment may compensate for reductions in external loading and phytoplankton biomass may not decline (Marsden 1989).

Nitrogen

Inlets, tiles, and wildlife pen sites Another important nutrient in aquatic systems is nitrogen. Total nitrogen consists of nitrate nitrogen, ammonia nitrogen, and organic nitrogen. Nitrate nitrogen made up almost all of the total nitrogen measured in Swan Lake inlets, tiles, and wildlife pen runoff (Table 11). Levels of ammonia nitrogen and organic nitrogen were low. High nitrate concentrations in runoff and tile effluent are typical in agricultural watersheds. Some of the sites in the Swan Lake watershed had very high concentrations of nitrate (Figure 10). Nitrate means calculated from all samples, including routine sampling and runoff events, show the inlet entering the lake from the southwest (INSW) had the highest concentrations with a mean above 25 mg/L. This site consistently had high concentrations of nitrate throughout the study period. The main west inlet (INW) had the second highest mean of 18 mg/L of nitrate. Other sites that were above 10 mg/L included wildlife pen number three (INWP3), the tile discharging into the south end of the lake (INS), and the tile discharging into the main west inlet (INCE).

Runoff events did not affect the levels of nitrate as they did phosphorus. Routine sample means were higher than runoff means of nitrate at six of the ten sites in the watershed (Table 12). This indicated nitrate levels where less variable between routinely taken samples and runoff events compared to phosphorus. Nitrate is readily leached from soil and this may account for the high concentrations measured throughout the study period compared to phosphorus that is usually bound to sediment transported in runoff.

Lake sites Concentrations of nitrate in the lake were much lower than concentrations measured in the inlets, tiles, and wildlife pen runoff. Nitrate that enters the lake becomes available for uptake by algae and is most likely assimilated. Levels of nitrate in the lake were higher in 1993 than levels measured in 1987-1989. In the post-restoration study over 80 percent of the nitrate samples were below the detection limit of 0.1 mg/L (IDNR 1993). Lake sites L1, L2, and L3 had similar levels of nitrate in 1993. The nitrate mean at site L1 was

Table 11. Means of total nitrogen, nitrate nitrogen, ammonia nitrogen, and organic nitrogen at Swan Lake inlets, tiles, and wildlife pen sites for all samples including routine sampling and runoff events in 1993

Site	Total Nitrogen (mg/L)			Nitrate Nitrogen (mg/L)		
	Sample Size	Mean	Standard Error	Sample Size	Mean	Standard Error
INW	25	19.804	1.298	24	18.216	1.334
INCE	21	13.396	2.692	22	10.932	2.430
INCW	8	3.544	1.392	8	2.294	1.245
INSW	25	27.916	1.546	25	25.625	1.414
INWP3	25	15.376	1.189	25	13.934	1.272
INS	23	13.770	1.195	24	12.721	1.327
INSE	23	10.373	1.088	24	8.918	1.214
INWP1	11	0.845	0.177	11	0.180	0.075
INN1	12	6.025	1.084	12	5.265	1.085
INN2	21	3.948	0.445	21	3.598	0.448

Site	Ammonia Nitrogen (mg/L)			Organic Nitrogen (mg/L)		
	Sample Size	Mean	Standard Error	Sample Size	Mean	Standard Error
INW	24	0.156	0.065	23	1.325	0.265
INCE	21	0.380	0.336	20	2.026	0.416
INCW	7	0.199	0.072	7	1.037	0.271
INSW	24	0.070	0.028	24	2.252	0.384
INWP3	24	0.175	0.140	24	1.213	0.224
INS	23	0.140	0.120	22	1.157	0.222
INSE	23	0.090	0.042	22	1.696	0.486
INWP1	11	0.172	0.063	11	0.494	0.112
INN1	12	0.283	0.135	12	0.478	0.129
INN2	21	0.094	0.050	21	0.255	0.055

1.605 mg/L, and 1.581 mg/L over all depths (Table 13). Lake sites L2 and L3 had means of 1.559 mg/L and 1.576 mg/L, respectively. When comparing levels of nitrate to the post-restoration study, the nitrate levels have apparently increased in the lake. Nitrate levels have likely increased due to increased amounts of precipitation in 1993. Fields in the watershed

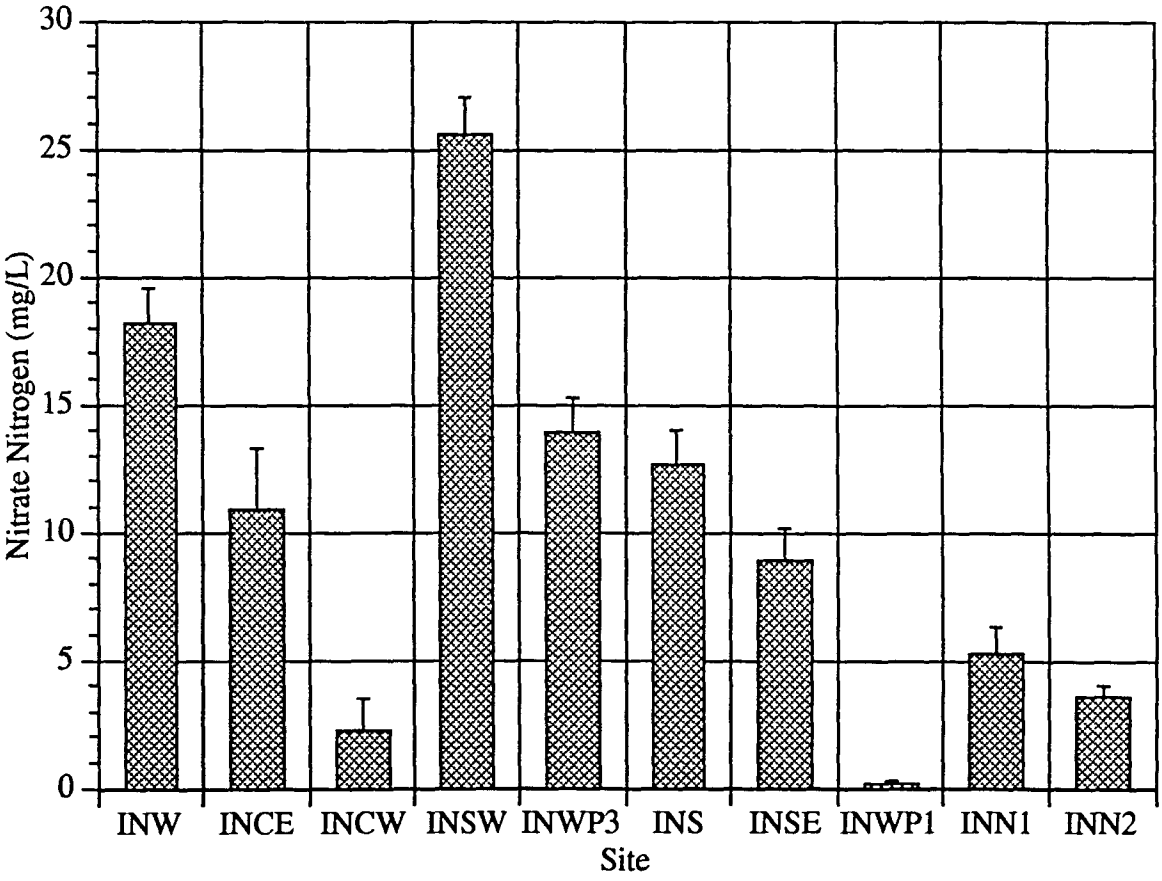

Figure 10. Nitrate nitrogen means with + one standard error in 1993 at Swan Lake inlets, tiles, and wildlife pen sites for all samples collected during both routine sampling and runoff events

were often saturated which would result in more nitrate leaching from the soil and being transported to the lake by inlets and tiles.

Suspended Solids

Inlets, tiles, and wildlife pen sites The sedimentation rate of Swan Lake is very low when compared to other lakes in Iowa. The sedimentation rate estimated by Bachmann et al. (1994) was only 0.8 cm per year; among the lowest in Iowa. The low sedimentation rate is

Table 12. Means of nitrate nitrogen (mg/L) and standard errors for routine sampling and runoff events at Swan Lake inlets, tiles, and wildlife pen sites in 1993

Samples taken during routine sampling

Site	Sample Size	Mean	Standard Error
INW	11	22.018	0.845
INCE	12	8.753	2.584
INCW	0		
INSW	12	29.445	0.485
INWP3	12	17.956	0.861
INS	12	14.986	1.661
INSE	12	10.905	1.803
INWP1	2	0.160	0.140
INN1	3	6.863	2.183
INN2	10	3.083	0.593

Samples taken during runoff events

Site	Sample Size	Mean	Standard Error
INW	13	14.998	1.977
INCE	10	13.548	4.374
INCW	8	2.294	1.245
INSW	13	22.099	2.308
INWP3	13	10.222	1.789
INS	12	10.457	1.917
INSE	12	6.932	1.480
INWP1	9	0.184	0.090
INN1	9	4.732	1.272
INN2	11	4.066	0.657

Table 13. Nitrate nitrogen means (mg/L) at various sites in Swan Lake in 1993

Site	Depth (m)	Sample Size	Mean	Standard Error
L1	0.5	13	1.605	0.199
L1	2.0	13	1.518	0.191
L1	3.5	13	1.620	0.237
L1	All	39	1.581	0.118
L2	0.5	13	1.559	0.203
L3	0.5	13	1.576	0.184

likely the result of a small watershed and implementation of soil conservation practices. In 1980 approximately 90 percent of the watershed was in approved soil conservation practices (Bachmann et al. 1982). However, in 1992 this percentage had dropped to only 50 percent (Bachmann et al. 1994).

Mean total suspended solids at Swan Lake inlets, tiles, and wildlife pen sites were low during routine sampling in 1993 (Figure 11). Only two sites, the northern inlet (INN1) and wildlife pen number one (INWP1), were above 10 mg/L and, as previously discussed, these sites were likely flowing during routine sample collection due to small rainfalls occurring prior to sampling. As expected, total suspended solid means were much higher during runoff events (Figure 12). The sites with the highest means included the tile that discharged into the main west inlet (INCE), wildlife pen number one (INWP1), site INCW located on the main west inlet above INCE, and the main west inlet (INW). Site INCE had the highest mean as a result of two runoff events on March 31 and August 30, 1993. Total suspended solid levels during these events were very high with 726 and 486 mg/L. Suspended solid levels at this site were relatively low during other events. When sites INWP1 and INCW were flowing they usually had high levels of suspended solids. Site INW is located on the largest inlet and had a mean of 54 mg/L.

Lake sites Comparison of data between 1980 and 1987-1989 showed similar levels of total suspended solids in Swan Lake. However, data from 1990 and 1993 show that levels of total suspended solids in the lake were much higher during those years (Figure 13). This may have been due to two factors. First, increased amounts of precipitation during these years may have increased the amount of suspended solids in the lake. Secondly, as indicated by the chlorophyll a means, algal biomass was higher during the years of 1990 and 1993 (Figure 14). Algal biomass would be represented in the organic portion of total suspended solids, and the organic portion was almost 60 percent of the mean total suspended solids at site L1 (0.5 m) in 1993 (Table 14).

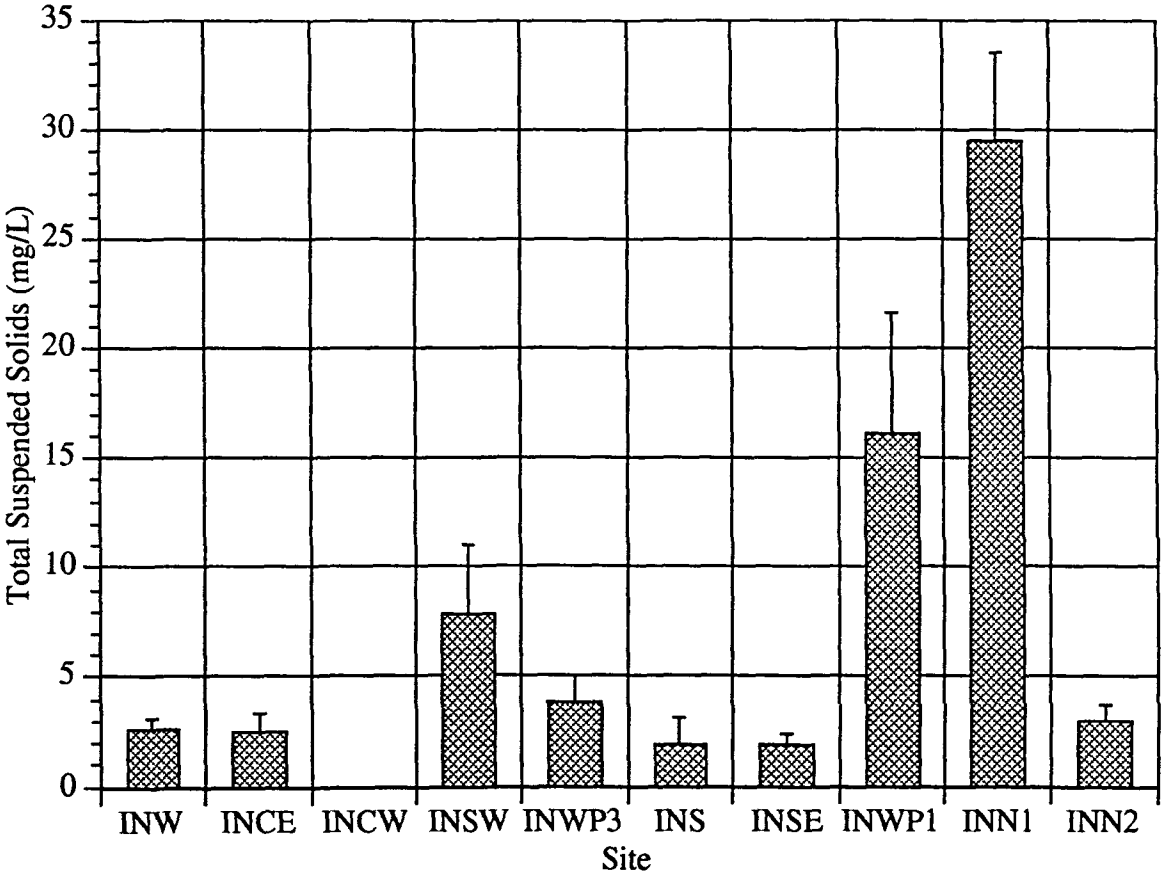

Figure 11. Total suspended solid means with + one standard error for Swan Lake inlets, tiles, and wildlife pen sites during routine sampling in 1993

The total suspended solid mean at lake site L1 was 34.8 mg/L ,and 34.3 mg/L over all depths (Table 14). Lake site L1 means were similar to lake sites L2 and L3 which had means of 36.2 mg/L and 36.3 mg/L of total suspended solids.

Chlorophyll a

Seasonal means of chlorophyll a at Swan Lake show most years were above 40 mg/m^3 (Figure 14). The 1990 and 1993 means were higher than previous years. Monthly means

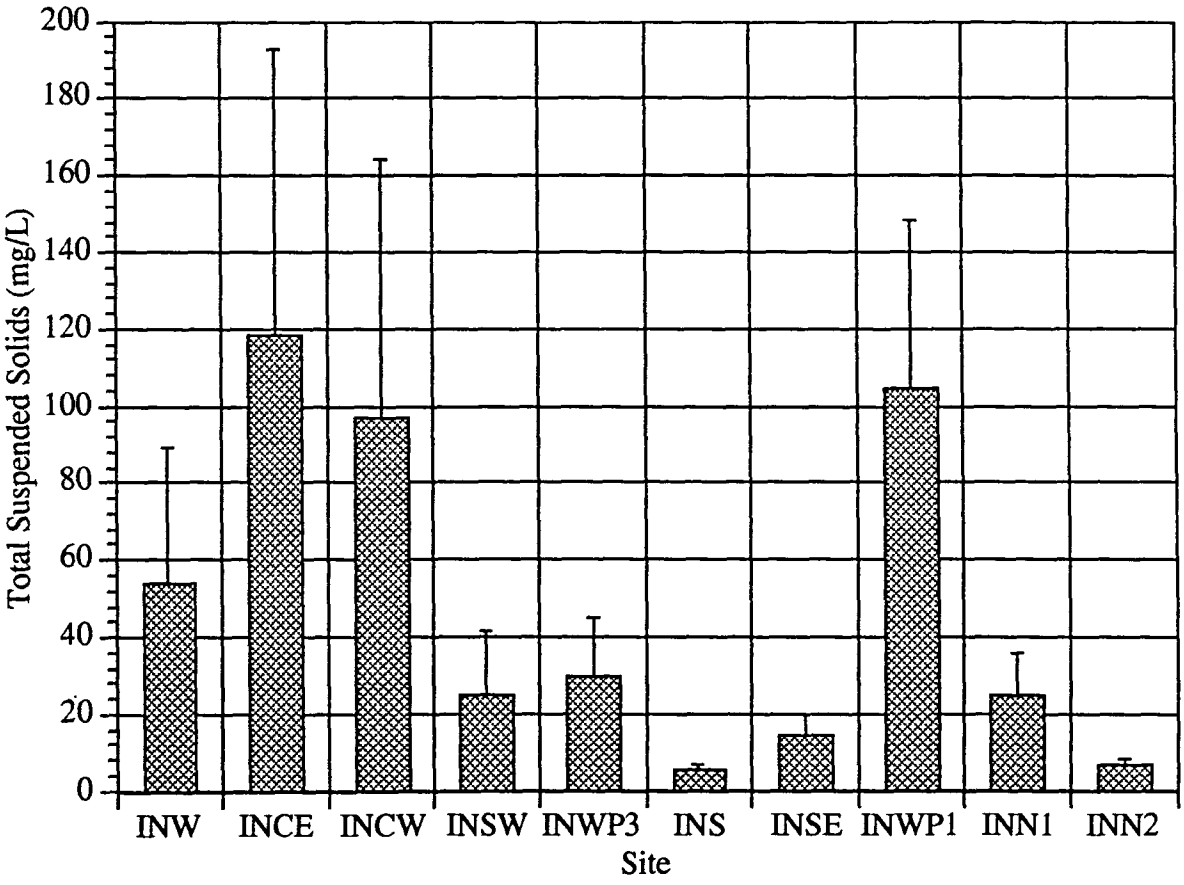

Figure 12. Total suspended solid means with + one standard error for Swan Lake inlets, tiles, and wildlife pen sites during runoff events in 1993

during the 1993 study period show June and August to have very high concentrations of chlorophyll a (Figure 15). The July mean was lower than June and August and this may have been due to the 18 cm of rain that fell on July 9, 1993. A rainfall of this magnitude would contribute high levels of total suspended solids to the lake and suppress the algal population through light limitation. On July 14, 1993 the chlorophyll a level at site L1 was only 35 mg/m^3, however by July 28, 1993 chlorophyll a levels were above 100 mg/m^3.

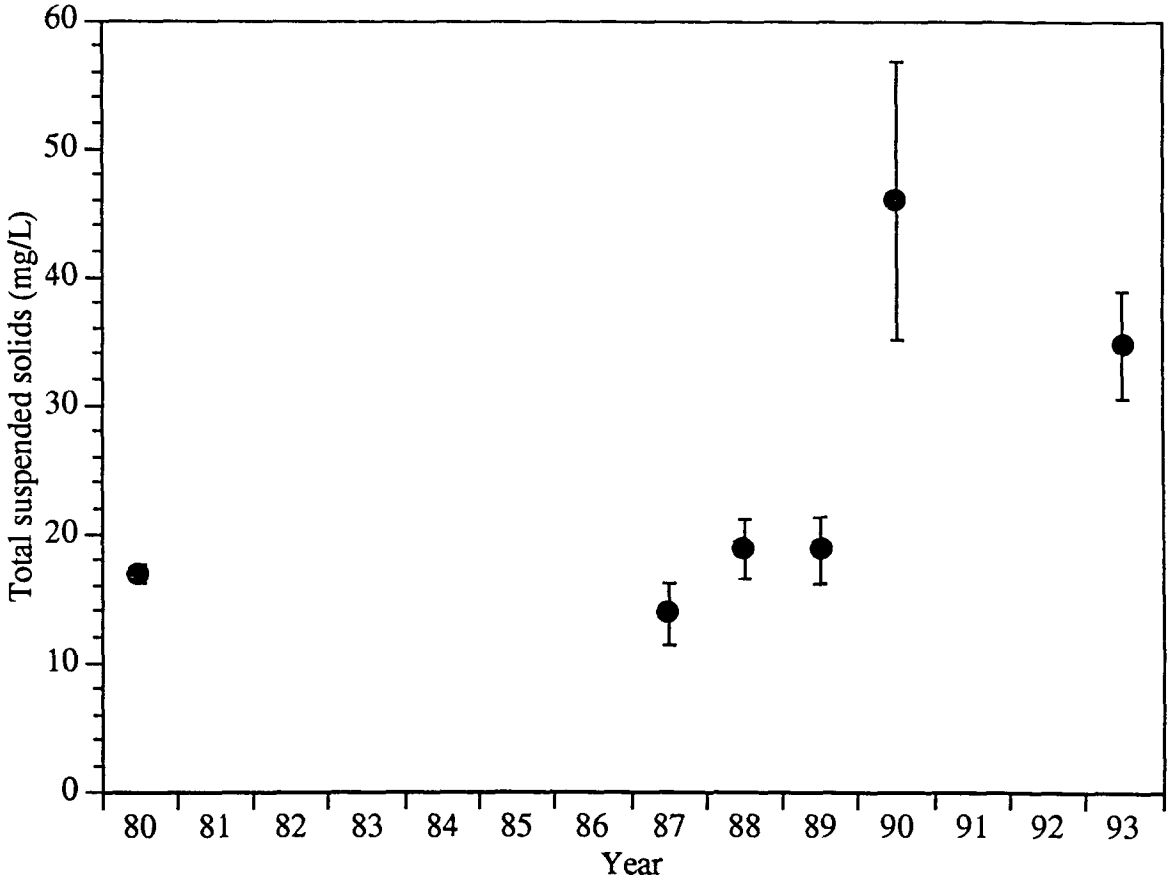

Figure 13. Seasonal means of total suspended solids at Swan Lake with +/- one standard error. Means are from the May-September period in 1980, 1987-1989, and 1993. The 1990 mean is from the June-August period. Means were calculated for 1980, 1987-1989, and 1990 from data collected by Bachmann et al. (1982), IDNR (1993), and Bachmann et al. (1994)

Chlorophyll a is used as an estimate of algal biomass. Comparison with previous years indicates algal blooms are still a very noticeable water quality problem at Swan Lake. Identification of algae genera showed most of the algae in 1993 consisted of blue-greens (Appendix C). The dominant genus throughout most of the study period was the blue-green

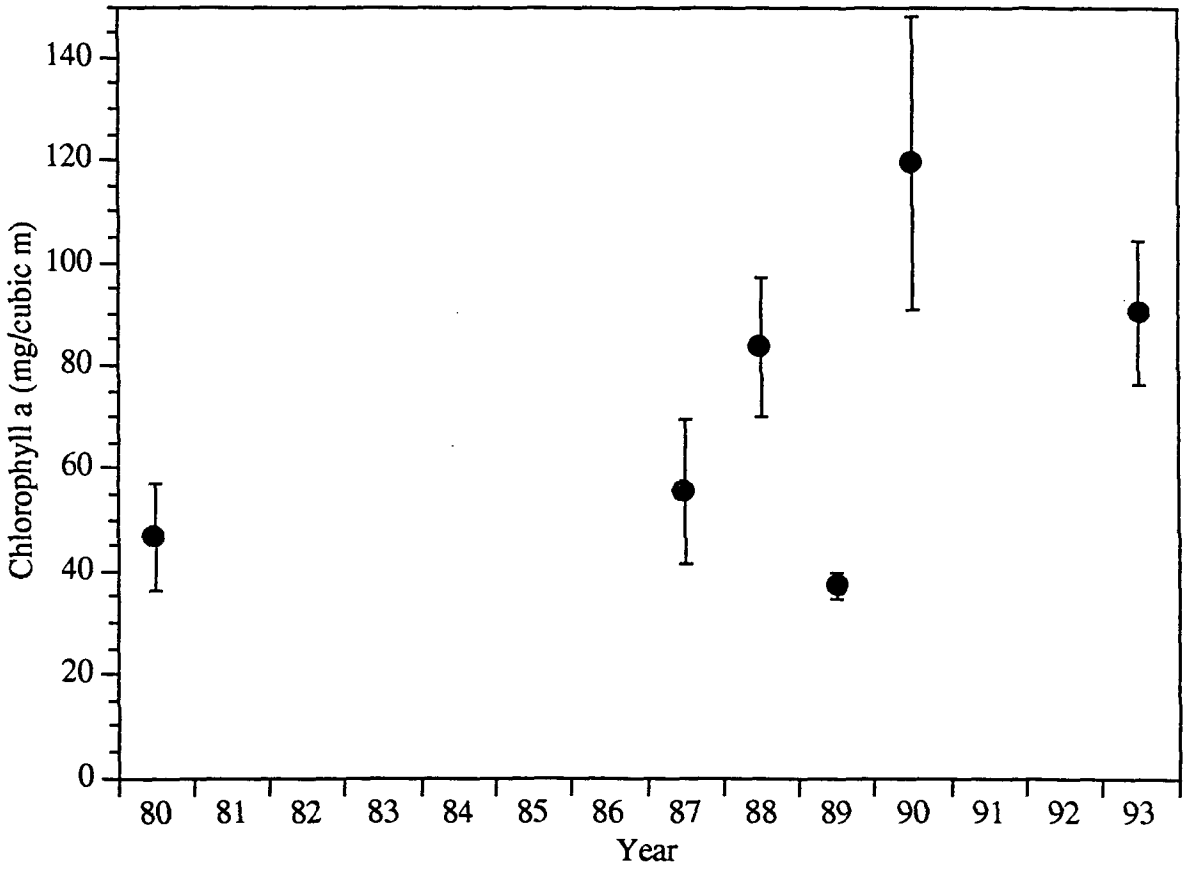

Figure 14. Seasonal means of chlorophyll a at Swan Lake with +/- one standard error. Means are from the May-September period in 1980, 1987-1989, and 1993. The 1990 mean is from the June-August period. Means were calculated for 1980, 1987-1989, and 1990 from data collected by Bachmann et al. (1982), IDNR (1993), and Bachmann et al. (1994)

alga *Anacystis*. Occasionally diatoms of genus *Stephanodiscus* contributed the largest cell volume.

The concentrations of nitrogen and phosphorus play a role in phytoplankton biomass and community structure. Phytoplankton are believed to be phosphorus limited in most temperate lakes; however, nitrogen can be the limiting nutrient in some systems (Schindler

Table 14. Total suspended solids, inorganic suspended solids, and organic suspended solid means at various sites in Swan Lake in 1993

Site	Depth (m)	Total Suspended Solids (mg/L) Sample Size	Mean	Standard Error	Inorganic Suspended Solids (mg/L) Sample Size	Mean	Standard Error
L1	0.5	13	34.769	4.237	13	14.571	2.045
L1	2.0	13	34.768	4.232	13	14.664	2.198
L1	3.5	13	33.498	3.656	13	15.258	2.155
L1	All	39	34.345	2.278	39	14.831	1.200
L2	0.5	13	36.203	4.683	13	14.051	2.085
L3	0.5	13	36.320	5.096	13	13.559	1.767

Site	Depth (m)	Organic Suspended Solids (mg/L) Sample Size	Mean	Standard Error
L1	0.5	13	20.198	2.863
L1	2.0	13	20.105	2.739
L1	3.5	13	18.240	2.327
L1	All	39	19.514	1.498
L2	0.5	13	22.152	3.564
L3	0.5	13	22.761	4.202

1977). The total nitrogen to total phosphorus ratio has been used to determine whether a lake is limited by nitrogen or phosphorus. In general, phosphorus limitation is expected when the nitrogen to phosphorus ratio is substantially larger than 15:1, and a ratio less than 15:1 suggests nitrogen limitation (Levine and Schindler 1992). Smith (1982) demonstrated phosphorus limitation occurs at ratios greater than 20:1. The nitrogen to phosphorus ratio for Swan Lake in 1993, calculated from the total nitrogen mean and the total phosphorus mean at site L1 over all depths, is 20:1. This indicates the lake is marginally phosphorus limited. At times the lake may be nitrogen limited, giving nitrogen-fixing blue-green algae an advantage over other taxa. Smith (1983) reported a tendency for blue-green algal blooms to occur when

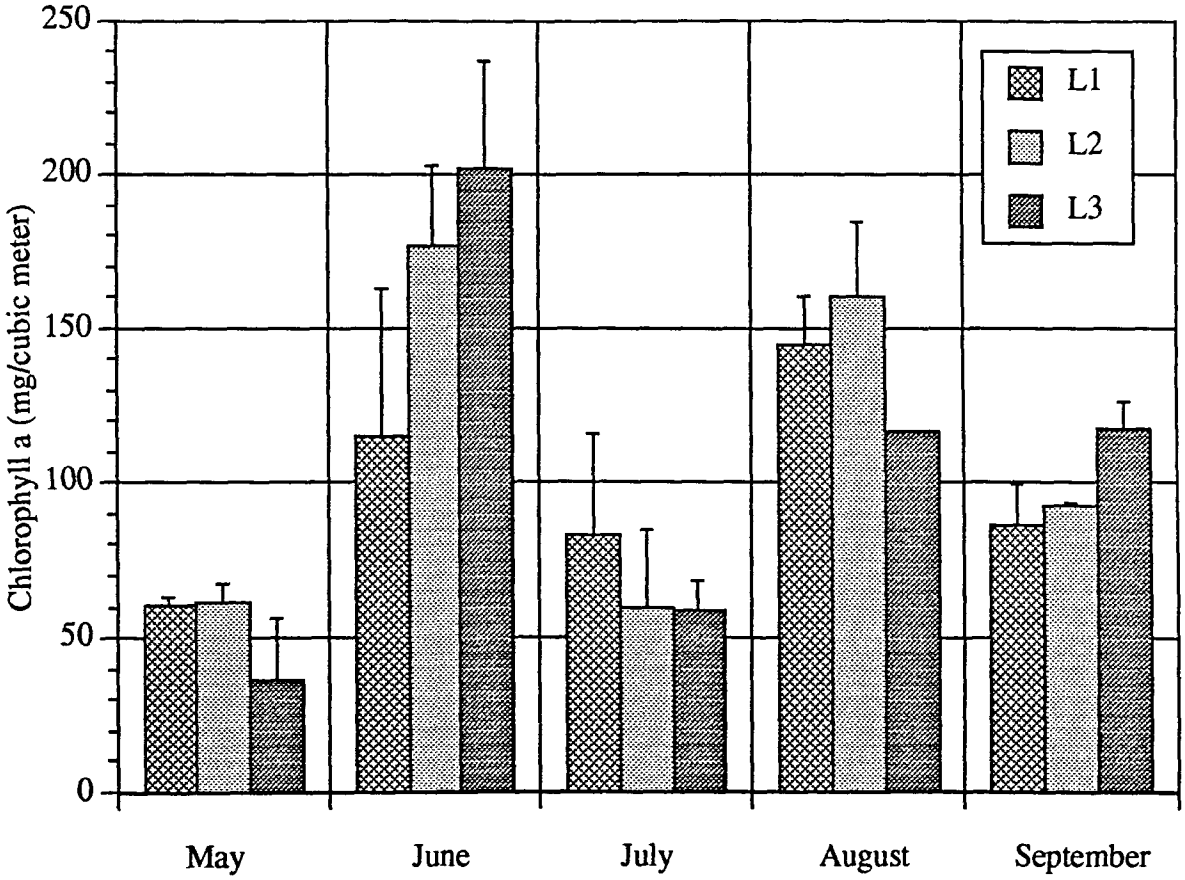

Figure 15. Monthly means of chlorophyll a with + one standard error from surface samples in 1993 at various sites in Swan Lake

nitrogen to phosphorus ratios fell below 29:1. Bachmann et al. (1982) also indicated both nitrogen and phosphorus most likely act as limiting nutrients at one time or another in Swan Lake. The nuisance algal blooms in Swan Lake are likely the result of large inputs of both nitrogen and phosphorus from the surrounding watershed.

Water transparency

Poor water transparency is a very noticeable water quality problem, and the public is concerned most with algal blooms and resulting low transparencies in Iowa lakes (Jones and Bachmann 1978). Poor water transparency was identified as one of the major water quality

problems at Swan Lake during the diagnostic feasibility study (Bachmann et al. 1982). The post-restoration study indicated water transparency had not improved following restoration (IDNR 1993).

Secchi disk depth was used as a measure of water transparency. Seasonal means of Secchi disk depth show most years had poor water transparency (Figure 16). The highest mean Secchi depth was in 1987; probably due to unusually high Secchi depths during May and June. This may have been related to decreased levels of suspended solids in the lake due to low amounts of precipitation during those months (IDNR 1993). The mean Secchi depth in 1990 was also better than other years, due to a high Secchi depth in June of that year. The Secchi depths in July and August of 1990 were much lower and more typical of other years at Swan Lake.

The mean Secchi disk depth at all lake sites in 1993 was only 0.4 m (Table 15). The 1993 mean is similar to most other years. Factors that influence water transparency include suspended solids and algal biomass. Chlorophyll a was used as an estimate of algal biomass and concentrations were quite high. Total suspended solids consist of organic and inorganic portions. Algal biomass is part of the organic matter in lakes, and a higher percentage of the total suspended solids consisted of the organic portion than the inorganic (Table 14). The total suspended solid mean was higher in 1993 than most previous years (Figure 13), but the chlorophyll a mean was also high in 1993 (Figure 14). Jones and Bachmann (1978) monitored 50 Iowa lakes and reservoirs and found poor water transparency was related more to algal biomass than to suspended inorganic matter. The poor water transparency at Swan Lake is more likely due to the high amount of algal biomass than inorganic suspended solids.

Dissolved Oxygen and Temperature

Restoration increased maximum lake depth from 2 to 4.5 m and the 1987-1989 post-restoration study (IDNR 1993) indicated thermal and chemical stratification occasionally occurred during the summer months. Data collected during the summer of 1993 show that the

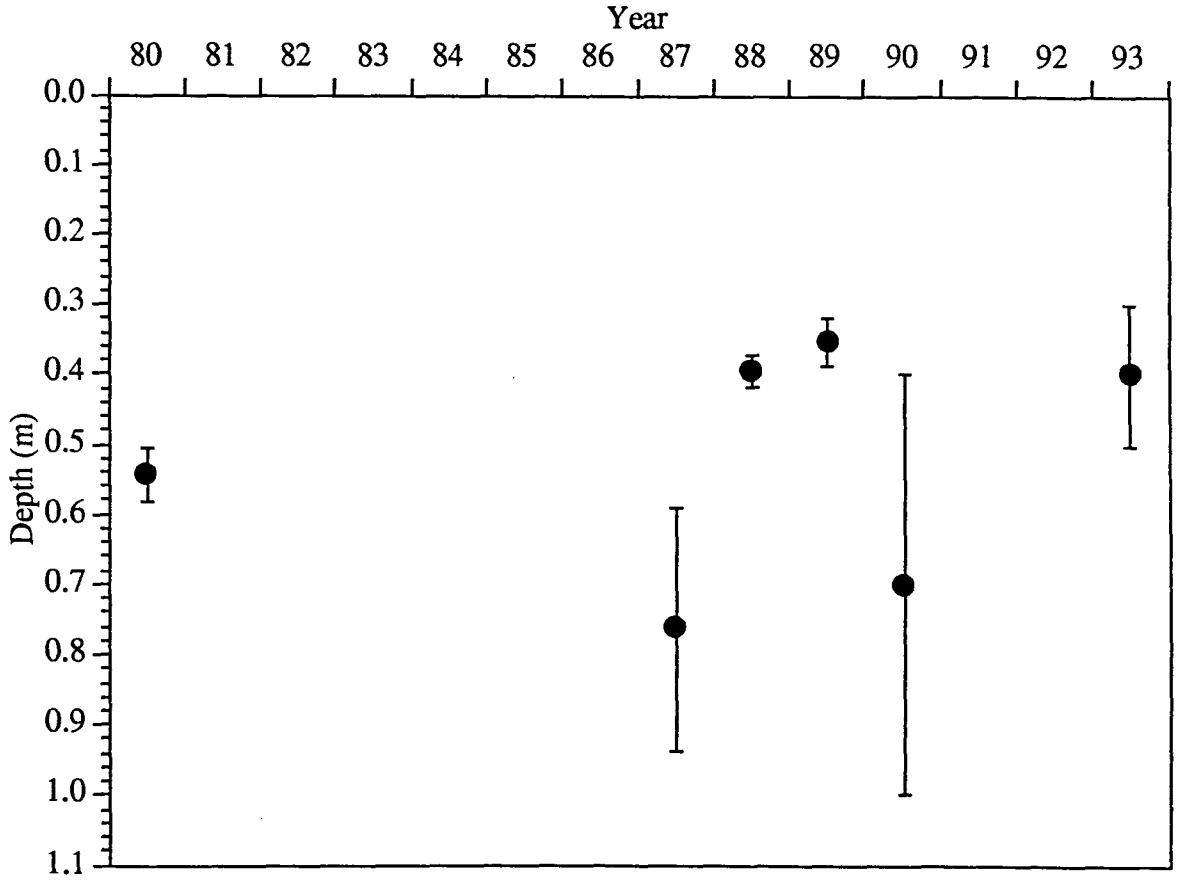

Figure 16. Seasonal means of Secchi disk depth at Swan Lake with +/- one standard error. Means are from the May-September period in 1980, 1987-1989, and 1993. The 1990 mean is from the June-August period. Means were calculated for 1980, 1987-1989, and 1990 from data collected by Bachmann et al. (1982), IDNR (1993), and Bachmann et al. (1994)

Table 15. Means of Secchi disk depth (m) at various sites in Swan Lake in 1993

Site	Sample size	Mean	Standard Error
L1	13	0.4	0.1
L2	13	0.4	0.1
L3	13	0.4	0.1

lake was weakly stratified on June 23 and July 14, 1993 at lake site L1 (Figure 17). Differences in dissolved oxygen between 0.5 m and 3.5 m on both dates were greater than 6.0 mg/L, and differences in temperature were close to 3 °C. Stratification may have occurred during windless periods at the deeper portion of the lake, but the lake was more typically mixed.

The mean dissolved oxygen at site L1 at 0.5 m below the surface was 11.0 mg/L, and 10.4 mg/L over all depths (Table 16). Lake site L2 had an mean of 11.2 mg/L, and lake site L3 11.7 mg/L. Dissolved oxygen levels were far above the requirements for aquatic organisms. The post-restoration study (IDNR 1993) indicated the installation of aeration equipment during lake restoration has increased dissolved oxygen levels during the winter months. Low dissolved oxygen levels in winter were a problem during some years prior to restoration as indicated by winter fish kills.

Alkalinity and pH

Means for total alkalinity, measured as mg/L as $CaCO_3$, and pH were calculated for lake sites L1, L2, and L3 (Table 17). The mean alkalinity at site L1 over all depths was 119 mg/L and the mean pH was 8.8. These means are typical of Iowa lakes (Bachmann 1965).

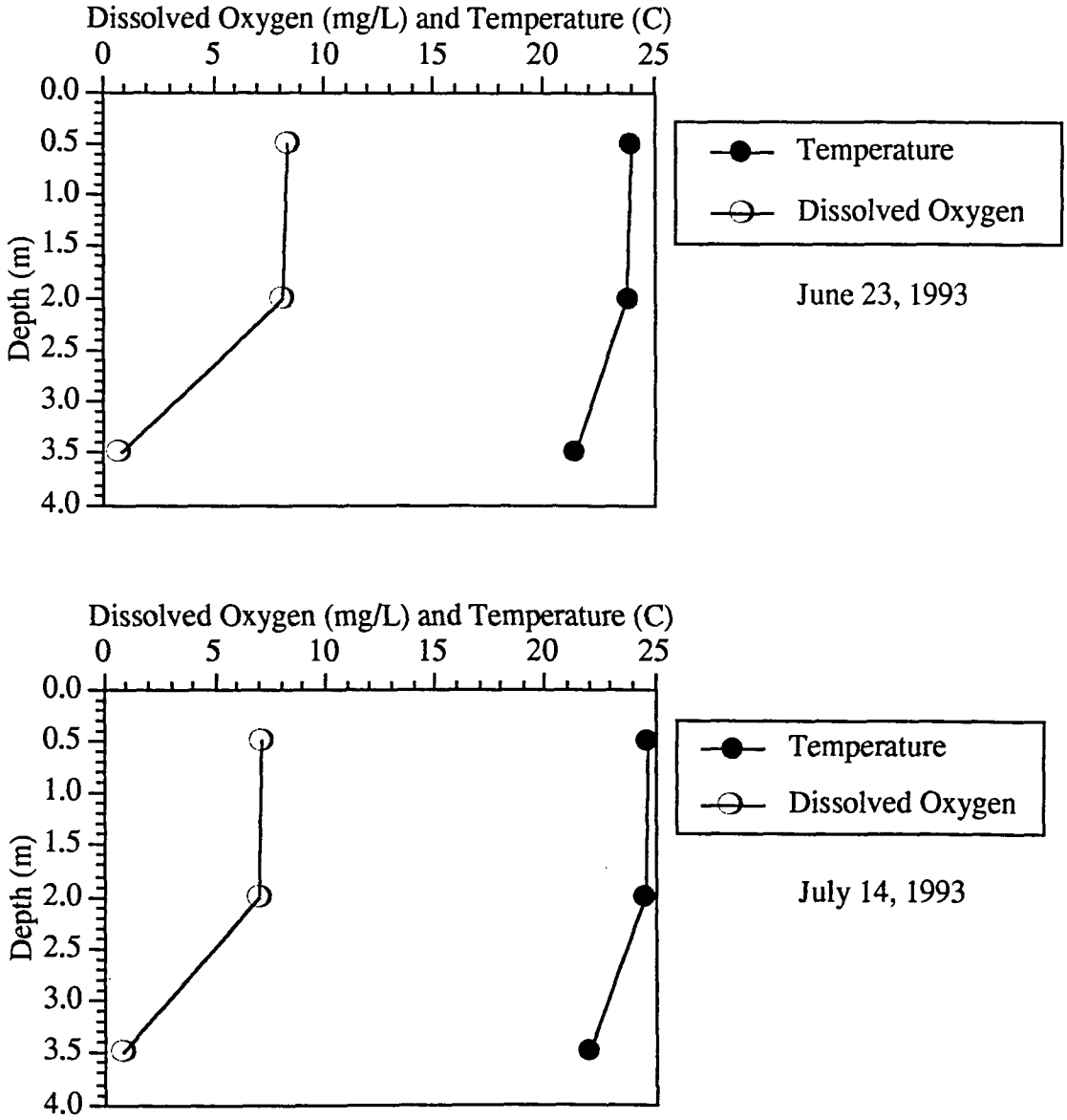

Figure 17. Swan Lake samples from site L1 showing most pronounced stratification in 1993

Table 16. Means of dissolved oxygen (mg/L) at various sites in Swan Lake in 1993

Site	Depth (m)	Sample Size	Mean	Standard Error
L1	0.5	13	11.0	0.7
L1	2.0	13	10.7	0.7
L1	3.5	13	9.5	1.2
L1	All	39	10.4	0.5
L2	0.5	13	11.2	0.7
L3	0.5	13	11.7	0.7

Table 17. Means of total alkalinity and pH at various sites in Swan Lake in 1993

		Total Alkalinity (mg/L as $CaCO_3$)			pH		
Site	Depth (m)	Sample Size	Mean	Standard Error	Sample Size	Mean	Standard Error
L1	0.5	11	118.5	5	13	8.9	0.1
L1	2.0	11	117.9	5	13	8.9	0.1
L1	3.5	11	120.3	5	13	8.8	0.1
L1	All	33	118.9	3	39	8.8	0.1
L2	0.5	11	118.9	5	13	8.9	0.1
L3	0.5	11	119.1	5	13	8.9	0.1

SUMMARY AND RECOMMENDATIONS

Bacterial contamination was greatly influenced by precipitation. Runoff events had significantly higher concentrations of bacteria than non-runoff conditions. Even though there was much variability between sites and between runoff and non-runoff conditions, some patterns existed by which the major sources of bacteria could be determined. The main west inlet (INW) and wildlife pen number three (INWP3) frequently had the highest concentrations of bacteria. Site INW drained agricultural land and bacteria entered this inlet from nonpoint runoff and from tile site INCE. Because INW and INWP3 were flowing throughout the study period, they may have consistently supplied bacteria to the lake. Wildlife pen number one (INWP1) required rain-induced runoff to be flowing, and had very high bacterial counts during runoff events. The August 30, 1993 runoff event had the greatest measured impact on water quality. During this event the highest fecal coliform bacteria concentration of the study (16 million colonies per 100 ml) measured at tile site INCE, also resulted in the highest bacterial counts measured in the lake. Animal waste from the cattle operation likely entered the tile system during this runoff event.

The measurements made in this study showed that fecal coliform bacteria levels often exceeded the water quality standard, particularly after periods of heavy rain. Bacteria entered the lake through inlets, tiles, and wildlife pen runoff. Cattle feedlots, wildlife housed in the pens, and manure spread on crop lands were most likely sources of this contamination. The fecal coliform bacteria water quality standard for primary contact recreation can likely be met under relatively dry weather conditions when runoff is not excessive. However, this standard will not likely to be met under prolonged wet weather conditions, especially under severe runoff conditions. Runoff induced by heavy rainfall leads to bacterial concentration increases resulting in water quality deterioration.

Total phosphorus means were shown to be high at most sites in the Swan Lake watershed during runoff events. Sites that had high means included INCW, INW, INSW, INCE, INWP1, and INWP3. Sites INCW, INW, and INSW drained nonpoint runoff from agricultural land in the watershed. Site INCE was tile effluent that also drained agricultural land. In addition, runoff from both wildlife pens had high concentrations of total phosphorus. The higher total phosphorus concentrations during runoff events are related to the higher total suspended solid concentrations measured during runoff events. It has been well established that phosphorus is transported in surface runoff adsorbed to suspended material. Total phosphorus samples collected during routine sampling had relatively low concentrations of total phosphorus when compared to runoff events.

Nitrate nitrogen made up almost all of the total nitrogen measured at sites in the watershed. Sites that had high means of nitrate included INW, INSW, INCE, INWP3, and INS. Sites INW and INSW drained nonpoint runoff, and sites INCE and INS were tile lines draining agricultural land. Site INWP3 contained drainage from wildlife pen number three. Concentrations of nitrate were less variable between routine sampling and runoff events.

The large inputs of both phosphorus and nitrogen maintain a total nitrogen to total phosphorus ratio of approximately 20:1 in Swan Lake. Both phosphorus and nitrogen may be the limiting nutrient at one time or another. The excess nutrient loading leads to nuisance algal blooms as indicated by high chlorophyll a values and poor water transparency.

To reduce the inputs of fecal coliform bacteria from the watershed improved animal waste management practices are needed. Because winter spreading of animal waste occurs in the watershed, any reduction in this practice should improve water quality. Waste application on frozen ground or snow cover should be avoided because the availability of bacteria to runoff would increase with no infiltration into frozen ground. If the application of animal waste must occur in the watershed, it should be plowed under to minimize the amount of waste entering runoff. Application should be avoided on natural drainage paths or waterways. Application

should also be avoided prior to a runoff event, however this practice is limited by the ability to predict the weather and the capacity of the operation. The extremely high bacterial count during the runoff event on August 30 at tile site INCE indicated strong point source fecal contamination and may have resulted from animal waste entering the tile system from the nearby cattle operation. This operation should locate manure storage areas away from hillsides or any other site with direct access to tile lines or drainage ditches. This would reduce the possibility of strong bacterial point-source contamination of nearby waterways in the watershed.

Other problem areas in the watershed include the wildlife pens located in the park. Water flowing from wildlife pen number three is likely needed to stabalize lake levels; therfore, animals in this pen may have to be removed to improve water quality. Runoff from wildlife pen number one flows across the road and into a small marshy area before entering the lake. It would seem reasonable to make the marshy area into a small lagoon to retain this runoff. Since this site requires rainfall to flow this structure would not have to be very large to capture and hold the runoff.

In addition to agriculture and wildlife pens, another possible source of fecal contamination may include park users. Additional studies should consider restrooms and associated septic tank lines in the park as a possible contributor of fecal contamination to the lake. The park has over 400,000 visitations annually and waste generated from park users should also be investigated.

Until reductions of fecal contamination can be achieved, swimming restrictions in Swan Lake may be necessary whenever an intense rainfall occurs. Bacteriological monitoring of the lake following a runoff event would determine when swimming could resume.

Improved nutrient management in the watershed may help reduce the inputs of phosphorus and nitrogen. In addition to reducing inputs of bacteria, careful management of animal waste would also reduce the concentration of nutrients transported in runoff. In 1990

approximately 50 percent of the watershed was in approved soil conservation practices (Bachmann et al. 1994). An increase in soil conservation practices may also reduce the amount of nutrients, such as phosphorus and ammonia nitrogen, that are transported in runoff attached to soil particles. Reductions in the amount of nitrogenous fertilizers applied to agricultural land in the watershed would also be beneficial.

The development of wetland areas bordering the lake may also help reduce the amount of nutrients entering the lake. Inlets and tiles flowing into wetland areas would be slowed and suspended material with attached phophorus would settle out. In addition to removal of phosphorus through sedimentation, denitrification in wetlands may also remove nitrate nitrogen.

Even if improvements in nutrient management are made, it is likely algal blooms will continue to be a water quality problem at Swan Lake. Nutrient inputs from the surrounding agricultural land would likely not be reduced by the magnitude needed to prevent algal blooms. This is a water quality problem that may have to be accepted at Swan Lake.

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APPENDIX A. LITERATURE REVIEW OF BACTERIAL INDICATORS

The following discussion reviews the use of bacterial indicators and the results of previous studies investigating fecal contamination in streams and lakes.

Bacterial Indicators

Bacterial concentrations in runoff from agricultural land become a concern when the land receives animal waste resulting from the production of livestock. Animal manure is applied to agricultural land by runoff from cattle feedlots, spreading manure on croplands, and grazing livestock. Runoff from agricultural land often reaches streams, rivers, and lakes that are used for recreational purposes such as swimming and water skiing. Concerns develop due to the potential for disease transmission by water-borne microorganisms of animal origin (Patni et al. 1985). Fecal pollution in runoff may add a variety of pathogens to water; the most common pathogens include strains of *Salmonella* (Geldreich 1972). The Iowa water quality standard to protect surface waters for primary contact recreation (Class A use) is 200 fecal coliform colonies per 100 ml of water sample, except when the waters are materially affected by surface runoff (IAC 1990). The risk of disease transmission has been found to increase substantially when fecal coliform counts exceed the water quality standard (Geldreich 1970).

The occurrence and density of pathogens in animals is highly variable, and monitoring water for specific pathogens requires elaborate and time consuming procedures. Therefore, bacterial indicators of fecal pollution are used. The higher the concentrations of bacterial indicators, the greater the probability of pathogens being present. Bacterial indicators are used to monitor fecal pollution of waters from both domestic and animal sources. The bacterial indicators most commonly used include total coliforms, fecal streptococci, and fecal coliforms. Studies may include all or one of these groups depending on the objectives of the research. Total coliforms are used to monitor treated waste water and older studies investigating nonpoint sources include total coliform data since water quality standards were first established for these indicators (Baxter-Potter and Gilliland 1988).

More recent studies monitoring water quality from nonpoint sources include data on fecal streptococci and fecal coliforms. Geldreich (1976) suggested the ratio of fecal streptococci to fecal coliform may be useful in identifying the source of fecal pollution. A ratio

of 4 or higher usually indicates contamination from human fecal material, and a range of 0.1 to 0.6 is typical of domestic animals. Values <0.1 are typical of wildlife. However, Geldreich (1976) cautions the ratio must be applied carefully because the ratio may change over time due to differences in die-off rates between the indicators, and once these organisms enter receiving streams various factors can influence the relationship. The use of fecal streptococci as an indicator has also been disputed due to different procedures and media used for identification (Baxter-Potter and Gilliland 1988). This makes comparisons of fecal streptococci data between studies difficult.

The fecal coliform has proven to be the most reliable indicator of fecal contamination. In aquatic systems where pollution originates from land runoff many coliform species of natural origin can be present making total coliforms insufficient as a measure of fecal contamination (Crane et al. 1983). Doran and Linn (1979) showed that high levels of total coliforms and fecal streptococci exist in agricultural runoff with or without animal waste applied to the land. Measures of fecal coliform eliminate this problem because fecal coliforms are only produced in the intestines of warm blooded animals (Crane et al. 1983). Fecal coliform bacteria have been found to have a direct correlation with fecal contamination from warm blooded animals (Geldreich 1972). They also have similar die-off rates as some pathogens, and have been correlated with an increase or decrease in pathogens such as *Salmonella* (Thornton et al. 1980). Several researchers agree that fecal coliform bacteria are the best indicator of fecal pollution (Geldreich 1965; Kunkle 1970; Doran and Linn 1979; Thelin and Gifford 1983; and Patni et al. 1985).

Several studies have been conducted using bacterial indicators in agricultural regions to monitor water quality for fecal pollution (Robbins et al. 1972; Buckhouse and Gifford 1976; Stephenson and Street 1978; Dudley and Karr 1979; Doran et al. 1981; Jawson et al 1982; Gary et al. 1983; Patni et al. 1984; and Patni et al. 1985). Results of these studies have shown bacterial indicators to be quite variable and influenced most by precipitation, waste

application methods, grazing activity, management practices of livestock operations, and contributions from wildlife. Valuable knowledge on bacterial indicators and fecal contamination can be gained by examining the results of previous studies.

The highly contaminated nature of feedlot runoff has led to investigations measuring concentrations of indicator bacteria in runoff from these sites. Rhodes and Hrubant (1972) measured concentrations of total coliforms in waste from a beef cattle feedlot near Peoria, Illinois. The feedlot waste contained 10^7 coliforms per gram dry weight. Concentrations varied only slightly during the study despite a wide variation in weather. Coliform concentrations were found to be extremely high in the feedlot waste stockpile and sites receiving runoff from the feedlot. Baxter-Potter and Gilliland (1988) identified concentrations of fecal coliform bacteria ranging from 1.3 million to 79 million colonies per 100 ml in cattle feedlot runoff. Results of these studies have shown runoff from feedlots can be a strong point source of fecal contamination. The concentration of bacterial indicators in feedlot runoff is about 10^4 times the concentration in runoff from land application sites (Khaleel et al. 1980).

The final management practice for feedlot operators is to apply the accumulated waste to cropland. Soils repeatedly exposed to animal waste contain varying amounts of bacterial indicators depending on weather conditions such as temperature and moisture (Van Donsel et al. 1967). Once animal waste is applied to the land it becomes a potential nonpoint source of pollution. Janzen et al. (1974) identified the potential for excess bacterial contamination in runoff from agricultural areas used for waste application. They surveyed streams for fecal coliform adjacent to South Carolina dairy farms using different manure application methods. Ninety percent of samples exceeded water quality criterion. The practice of solid disposal of waste was found to contribute more fecal pollution to runoff than application of liquid manure. Crane et al. (1983) suggested that the most important factor in controlling transport of microorganisms from waste application sites is the contact time with soil prior to a rainfall event. Liquid waste comes into immediate contact with soil following application and this may

reduce surface losses as long as the application rate does not create runoff conditions. Solid waste applied on the surface is readily exposed to rainfall increasing contamination in runoff.

Patni et al. (1985) measured concentrations of indicator bacteria in runoff from adjacent manured and non-manured watersheds in Ontario, Canada. Significant differences in quality of runoff from manured and non-manured croplands were not always observed under heavy runoff conditions. Heavy runoff under wet weather conditions was found to degrade water quality regardless of manuring activity. When compared to results of other studies, runoff from the manured watershed was of better quality, and this was attributed to the management practice of dry weather manure application followed by plowdown. They also measured much lower indicator bacteria in long-term stored manure than in relatively fresh manure. This indicated a lower potential for fecal pollution from application of long-term stored manure.

In addition to direct application of animal waste to agricultural land, grazing livestock can also be a potential nonpoint source of fecal pollution. Doran et al. (1981) found cattle grazing resulted in a 5 to 10 fold increase in fecal coliform counts in runoff in eastern Nebraska. However, fecal coliform counts in rainfall runoff from both grazed pasture and an ungrazed control area exceeded water quality criteria more than 90 percent of the time. They concluded fecal contributions from wildlife can have a substantial effect on the quality of pasture runoff and other areas where wildlife are found.

Total coliform, fecal coliform, and fecal streptococci concentrations were monitored by Jawson et al. (1982) for 3 years to determine the effect of grazing on the presence of these organisms in runoff from a grazed and nongrazed watershed in the Pacific Northwest. Average numbers of total coliform and fecal streptococci in runoff did not differ significantly, but fecal coliform concentrations were found to be higher from the grazed area. There was some correlation between recentness of grazing and numbers of indicator bacteria in runoff. However, more than a year after animals were removed from the watershed fecal coliform numbers in runoff still exceeded 200 colonies per 100 mL, and not until two years later did

they drop to <10 colonies per 100 mL. Boyer and Perry (1987) also showed the effects of recentness of grazing activity on fecal coliform concentrations in runoff from a reclaimed surface mine in southern West Virginia. Prior to grazing, fecal coliform counts were <20 colonies per 100 ml, and one year after grazing began bacteria concentrations were >2500 colonies per 100 ml. After grazing ceased fecal coliform counts remained high for several months. Stephenson and Street (1978) also found fecal coliform counts to remain high for several months after cattle were removed from a watershed in southwest Idaho.

Gary et al. (1983) found the density of cattle grazing to have an effect on indicator bacteria concentrations in pastures bisected by a small stream in central Colorado. Indicator bacteria concentrations in the stream were significantly higher when at least 150 cattle were grazing. When <40 cattle were grazing, bacteria concentrations dropped to levels of those in an adjacent ungrazed pasture. They also found about 5 percent of the total manure produced by cattle contributed to pollution of the stream. Milne (1976) also observed significantly higher bacteria concentrations in a stream located near the greatest livestock grazing activity compared to concentrations upstream. Upstream counts of fecal coliform bacteria averaged 18 colonies per 100 ml, whereas downstream from the grazing area counts averaged 997 colonies per 100 ml.

During the summers of 1973 and 1974 a water quality study was conducted on a relatively dry rangeland in southeastern Utah (Buckhouse and Gifford 1976). Comparison of grazed and ungrazed watersheds showed no significant changes in indicator bacteria from grazing use. It was suggested this may have been due to dry rangeland conditions, gently sloping watersheds, and a low grazing use rate. The potential public health hazard of livestock grazing on semiarid rangeland was minimal.

Studies have also monitored concentrations of bacterial indicators in watersheds with multiple farming activities contributing to fecal contamination. Robbins et al. (1972) investigated agricultural sites in North Carolina. Five of the six sites were watersheds

subjected to animal waste application, pastures, and feedlot operations. A sixth watershed free of animal waste was used as a control. Samples were analyzed for total coliforms, fecal streptococci, and fecal coliforms. The authors observed a strong relationship between bacteria concentrations and storm events. The bacterial quality in streams in all six watersheds greatly exceeded water quality standards at higher stream discharge. Since bacteria concentrations from the control watershed were also high they suggested other factors such as rainfall, temperature, slope of the watershed, and degree of erosion may influence bacteria concentrations.

Fecal contamination of an agricultural drainage in northeast Indiana was also found to be greatest during high stream discharge following storm events (Dudley and Karr 1979). Samples were analyzed for total coliforms, fecal streptococci, and fecal coliforms. The sources of contamination were determined to be from septic tanks and livestock operations. Sections of the drainage received septic effluent and had coliform counts far in excess of public health standards during low and high flow. During storm runoff events fecally contaminated sediment was transported from unconfined livestock operations. Areas of the drainage distant from septic tank pollution were found to occasionally meet water quality standards but generally these areas had twice the allowable limit.

Several additional studies have shown concentrations of indicator bacteria in runoff from agricultural lands to exceed water quality standards (Kittrell and Furfari 1963; Weidner et al. 1969; Barker and Sewell 1973; Doran and Linn 1979; Khaleel et al. 1980; and Niemi and Niemi 1991). This is true of virtually all agricultural land uses in most regions. The water quality standards were initially developed for point sources such as water treatment plants. Because bacterial concentrations in agricultural runoff seldom meet water quality standards there has been some discussion on whether the water quality standards developed for waters receiving point sources are also applicable to waters under the influence of nonpoint sources (Doran and Linn 1979; and Doran et al. 1981).

Because runoff from agricultural land often exceeds water quality standards for primary contact recreation, the water quality of lakes receiving runoff becomes a concern due to public health hazards associated with fecal contamination. Studies have also monitored the transport of bacterial indicators in runoff and their distribution in lakes.

Geldreich (1972) monitored recreational water quality in Buffalo Lake near Amirillo, Texas. The Buffalo Lake watershed drained pastures, cultivated farmland, cattle feedlot operations, and sewage treatment plant effluent. During long dry periods the bathing water quality in the lake was found to meet water quality standards for fecal coliform bacteria. Runoff produced by heavy rains resulted in dramatic increases in bacteria concentrations in feeder streams. Following runoff events concentrations of fecal coliform bacteria in the lake were 240,000 per 100 ml, far in excess of the 200 per ml water quality standard. The distribution of bacteria in Buffalo Lake showed a decrease in concentrations at the lower third of the lake nearest the dam. It was suggested that dilution and sedimentation of bacteria played a role in the distribution.

Thornton et al. (1980) also found dilution and sedimentation to affect bacterial distribution. They sampled storm events in the Caddo River and in DeGray Reservoir, Arkansas. Storm events were tracked through the reservoir using increased turbidity associated with storm events. Fecal coliforms were sampled in the river and throughout the water column in the reservoir. Increased fecal coliform concentrations were closely associated with increased turbidity from runoff events. As the turbidity plume moved down the reservoir fecal coliform concentrations decreased due to sedimentation and dilution. Gannon et al. (1983) observed a similar pattern in Ford Lake, Michigan. Sedimentation was demonstrated as important in the disappearance of fecal coliform bacteria in the lower end of the lake.

Pugh and Reyes (1991) monitored seven recreational areas on Percy Priest Reservoir, Tennessee. They measured fecal streptococci and fecal coliform from May through August, 1986, and found that the reservoir was safe for primary contact recreation throughout the

summer. Sekla et al. (1987) identified 15 beaches in Manitoba, Canada that met water quality criteria for fecal coliform during the summers of 1984 and 1985.

It is evident that bacterial concentrations in runoff from agricultural land will be reduced by dilution once runoff enters a receiving body of water used for recreational purposes. Whether or not the receiving body of water exceeds water quality standards will depend most on land use practices in the watershed, precipitation, and the size of the receiving water body.

APPENDIX B. LITERATURE REVIEW OF PLANT NUTRIENTS

The following discussion reviews the literature on plant nutrients, nitrogen and phosphorus, and their effect on lake water quality.

Plant Nutrients

Lake primary productivity is largely determined by the concentrations of nutrients entering through surface runoff, precipitation, and groundwater drainage. Nitrogen and phosphorus are both recognized as limiting nutrients in aquatic systems. Because most phosphorus is adsorbed to soil particles there is little movement of this nutrient in subsurface flow or groundwater. However, inorganic forms of nitrogen are soluble in water and are transported in subsurface flow. Inputs of nitrogen and phosphorus from precipitation are highly variable and can be influenced by many factors including geographic location and weather patterns. In general, inputs of nutrients from precipitation are considered minor compared to inputs from terrestrial runoff (Wetzel 1983).

Lakes that have high levels of nutrients and productivity are classified as eutrophic. Nutrient enrichment of lakes caused by human activities is referred to as "cultural eutrophication" (Bachmann 1980). This is apparent in Iowa where lakes typically receive an excess of nutrients transported in runoff from agricultural land. This can result in high densities of algae or aquatic plants, poor water transparency, reduced dissolved oxygen in the hypolimnion of stratified lakes from decomposing organic matter, and in shallow lakes dissolved oxygen under the ice in winter may be depleted below requirements of fish resulting in fishkills.

The role of nitrogen and phosphorus in lake productivity was demonstrated by experimental lake studies in Canada (Schindler et al. 1973 and Schindler 1974). Oligotrophic lakes were used in experiments where various levels of nitrogen and phosphorus were added. After the addition of nutrients they found algal standing crops increased almost two orders of magnitude, similar to algal crops in eutrophic lakes. In addition, both phosphorus and nitrogen were required to produce large standing crops of phytoplankton, but the addition of nutrients singly revealed that phosphorus always caused some increase in standing crop, while addition of nitrogen alone evoked no response.

Reduction in lake productivity by decreasing phosphorus inputs was demonstrated in Lake Washington, Seattle (Edmondson 1971). The lake was enriched with effluent from sewage treatment plants from 1941-1963, and phosphorus concentrations were found to increase much more than nitrogen. During this time production increased and algae became more abundant. After diversion of effluent both phosphorus concentrations and algae decreased in the lake, while nitrogen fluctuated between years at relatively high levels indicating the importance of phosphorus to phytoplankton.

Studies have also shown strong correlations between phosphorus and chlorophyll a concentrations in diverse groups of lakes (Dillon and Rigler 1974 and Jones and Bachmann 1976). Bachmann and Jones (1974) established relationships between total phosphorus, chlorophyll a, and Secchi disk depth. These relationships showed that phosphorus inputs would have to be reduced to extremely low levels of about 0.02 mg/L before significant increases in water transparency would be achieved as a result of reduced algal crops. Results from these studies have also shown the importance of phosphorus on algal densities in lakes.

Most temperate lakes are believed to be phosphorus limited (Schindler 1977), however Elser et al. (1990) suggested nitrogen may have a more important role than previously recognized and greater attention should be given to both phosphorus and nitrogen. Although both nitrogen and phosphorus are recognized as limiting nutrients, most studies have focused on phosphorus because previous studies have shown the importance of phosphorus, and the ability of some blue-green algae to fix atmospheric nitrogen may make up for nitrogen deficiencies (Bachmann 1980).

Reduction of external phosphorus loading is usually the greatest concern in lake restoration programs because of the importance of phosphorus as a limiting nutrient and its chemical nature (Bachmann 1980). It has been shown that phosphorus is transported from watersheds adsorbed to soil particles in surface runoff (Taylor 1967 and Timmons 1968). Therefore, it should be possible to reduce inputs from watersheds by soil conservation

practices that reduce erosion. However, inorganic forms of nitrogen are soluble in water, and nitrogen can be fixed from the atmosphere by blue-green algae making nitrogen control less feasible.

Point sources of nutrients, such as wastewater effluent, are relatively easy to control compared to nonpoint sources from agricultural land where fields are applied with fertilizers containing nitrogen and phosphorus. High percentages of nutrients originating from fertilizer are often lost to surface runoff (Kohl et al. 1971). In addition to surface runoff, Jones et al. (1976) found tile lines draining agricultural land in Iowa transport high concentrations of nitrate nitrogen. Researchers investigating nutrient losses from agricultural land have shown losses to depend on erosion, precipitation, soil characteristics, watershed characteristics, and soil conservation practices (Taylor 1967; Timmons et al. 1968; Romkens et al. 1973; Burwell et al. 1974; and Burwell et al. 1977).

Animal waste resulting from the production of livestock has also been identified as a source of phosphorus and nitrogen in agricultural watersheds. Brown et al. (1989) monitored runoff from feedlots and manure-spread cropland in the watershed of Cannonsville Reservoir, New York. They found that runoff from the watershed during the winter-spring period accounted for approximately 80 percent of the annual loading of total phosphorus to the reservoir. The spring phosphorus concentration largely determined the extent of the algal biomass during the summer. The reservoir was described as eutrophic with algal blooms that substantially reduced water transparency.

Klausner et al. (1976) found that manure spreading during thaw periods in spring resulted in increased nutrient losses of nitrogen and phosphorus in a New York watershed. Young and Mutchler (1976) also showed the effect of snow melt runoff from manure applied to frozen ground in Minnesota. Concentrations of nitrogen and phosphorus were found to be high from all manured areas compared to unmanured areas.

Jones et al. (1976) measured nutrient loads of tributary streams of lakes in northwestern Iowa. Watershed land-use practices were used to determine differences in nutrient loads between streams. Livestock were an identifiable source of phosphorus and ammonia nitrogen. Animal units in feedlots were significantly correlated with these nutrients.

Quantifying inputs of nutrients to lakes from nonpoint sources, such as croplands and feedlots, is an important step in attempting to control eutrophication. Inputs from point sources are relatively easy to estimate, however inputs from nonpoint sources are difficult to estimate because of variation in weather and associated runoff events. When considering the amount of time, work, and cost involved in making field measurements of nutrient budgets, it is impossible to take measurements directly when planning restoration for a large number of lakes (Dillon and Kirchner 1975). Therefore, models of nutrient loading to lakes have been developed (Dillon and Rigler 1974 and Bachmann 1984). Information from models should be beneficial to lake managers in determining the expected degree of eutrophication and possible benefits of restoration practices.

While removal of point-source pollution to a lake can be beneficial, less is known about the benefits from reducing nonpoint sources of nitrogen and phosphorus. Reducing nutrient loads from nonpoint sources may improve water quality in some lakes. The extent of improvement of an individual lake will likely depend on the degree of eutrophication and its history of nutrient loading. Studies have indicated large reductions in nutrients would be required to achieve noticeable changes in lake water quality (Bachmann and Jones 1974 and Marsden 1989).

APPENDIX C. ORIGINAL DATA COLLECTED AT SWAN LAKE IN 1993

The following tables contain original data on measurements of fecal coliform bacteria, Secchi disk depth, dissolved oxygen, temperature, pH, total alkalinity, chlorophyll a, total phosphorus, soluble reactive phosphorus, total suspended solids, inorganic suspended solids, organic suspended solids, total nitrogen, nitrate nitrogen, ammonia nitrogen, organic nitrogen, and algae biovolumes.

Table C1. Measurements of fecal coliform bacteria at Swan Lake inlets, tiles, wildlife pens, and lake sites in 1993

Date	Site	Fecal Coliform Colonies/100 ml
6/2/93	L1	78
	L2	56
	L3	20
	L4	34
	L5	58
	INW	75
	INCE	1
	INCW	
	INSW	84
	INWP3	174
	INS	10
	INSE	127
	INWP1	1780
	INN1	
	INN2	208
	Big Creek	170
6/9/93	INW	169
	INCE	5
	INCW	
	INSW	19
	INWP3	115
	INS	3
	INSE	22
	INWP1	
	INN1	
	INN2	
	Big Creek	188
6/14/93	INW	910
	INCE	13
	INCW	1160
	INSW	66
	INWP3	1025
	INS	27
	INSE	89
	INWP1	
	INN1	590
	INN2	395
6/23/93	INW	445
	INCE	2
	INCW	
	INSW	40
	INWP3	535

Table C1. Cont.

Date	Site	Fecal Coliform Colonies/100 ml
6/23/93 Cont.	INS	3
	INSE	
	INWP1	
	INN1	
	INN2	
	Big Creek	94
6/29/93	L1	14
	L2	220
	L3	17
	L4	300
	L5	290
6/30/93	INW	2200
	INCE	69
	INCW	
	INSW	1750
	INWP3	4200
	INS	54
	INSE	536
	INWP1	41000
	INN1	
	INN2	3250
7/1/93	INW	655
	INCE	216
	INCW	
	INSW	518
	INWP3	3610
	INS	
	INSE	525
	INWP1	11525
	INN1	
	INN2	12850
7/9/93	INW	32500
	INCE	
	INCW	3850
	INSW	13500
	INWP3	3450
	INS	
	INSE	
	INWP1	41500
	INN1	1870
	INN2	

Table C1. Cont.

Date	Site	Fecal Coliform Colonies/100 ml
7/14/93	L1	155
	L2	300
	L3	140
	L4	900
	L5	500
	INW	291
	INCE	1065
	INCW	
	INSW	362
	INWP3	729
	INS	46
	INSE	212
	INWP1	
	INN1	640
	INN2	672
	Big Creek	210
7/28/93	L1	140
	L2	280
	L3	220
	L4	195
	L5	110
	INW	349
	INCE	87
	INCW	
	INSW	376
	INWP3	450
	INS	30
	INSE	350
	INWP1	143
	INN1	1930
	INN2	1810
	Big Creek	630
8/9/93	L1	100
	L2	140
	L3	
	L4	170
	L5	220
	INW	442
	INCE	17
	INCW	
	INSW	162
	INWP3	
	INS	5
	INSE	230

Table C1. Cont.

Date	Site	Fecal Coliform Colonies/100 ml
8/9/93 Cont.	INWP1	
	INN1	312
	INN2	970
	Big Creek	43
8/29/93	INW	11450
	INCE	6050
	INCW	14950
	INSW	2815
	INWP3	9200
	INS	792
	INSE	475
	INWP1	52000
	INN1	
	INN2	6200
8/30/93	L1	90
	L2	800
	L3	3000
	L4	5000
	L5	900
	INW	810000
	INCE	16000000
	INCW	
	INSW	7350
	INWP3	10300
	INS	503
	INSE	3825
	INWP1	48500
	INN1	4000
	INN2	2150
9/14/93	L1	40
	L2	130
	L3	140
	L4	170
	L5	110
	INW	379
	INCE	670
	INCW	
	INSW	200
	INWP3	203
	INS	10
	INSE	109
	INWP1	
	INN1	

Table C1. Cont.

Date	Site	Fecal Coliform Colonies/100 ml
9/14/93 Cont.	INN2	145
	Big Creek	310
9/26/93	L1	20
	L2	2
	L3	20
	L4	20
	L5	40
	INW	80
	INCE	33
	INCW	
	INSW	182
	INWP3	115
	INS	9
	INSE	235
	INWP1	
	INN1	
	INN2	55
	Big Creek	385
10/12/93	L1	0
	L2	2
	L3	2
	L4	2
	L5	2
	INW	28
	INCE	32
	INCW	
	INSW	103
	INWP3	287
	INS	1
	INSE	26
	INWP1	
	INN1	
	INN2	2
	Big Creek	124

Table C2. Measurements of Secchi disk depth, temperature, and dissolved oxygen in 1993 at various sites in Swan Lake

Date	Site	Depth (m)	Secchi (m)	Temp. (°C)	D.O. (mg/l)
4/27/93	L1	0.5	0.6	13.1	11.6
	L1	2.0		13.1	11.7
	L1	3.5		13.1	11.6
	L2	0.5	0.5	13.3	12.5
	L3	0.5	0.5	13.3	12.2
5/19/93	L1	0.5	0.5	16.7	11.4
	L1	2.0		16.8	11.4
	L1	3.5		16.8	11.4
	L2	0.5	0.6	16.6	11.6
	L3	0.5	0.6	16.9	13.0
5/26/93	L1	0.5	0.5	18.2	14.5
	L1	2.0		16.8	12.8
	L1	3.5		16.4	12.2
	L2	0.5	0.4	18.3	14.6
	L3	0.5	0.4	18.0	14.8
6/2/93	L1	0.5	0.3	15.7	12.0
	L1	2.0		15.8	12.0
	L1	3.5		15.8	12.0
	L2	0.5	0.4	15.4	12.1
	L3	0.5	0.4	15.3	12.3
6/9/93	L1	0.5	0.2	19.6	13.1
	L1	2.0		19.5	12.8
	L1	3.5		19.4	12.5
	L2	0.5	0.2	19.7	13.1
	L3	0.5	0.2	19.8	13.3
6/23/93	L1	0.5	0.2	24.0	8.4
	L1	2.0		23.8	8.2
	L1	3.5		21.4	0.8
	L2	0.5	0.2	24.9	9.7
	L3	0.5	0.2	25.8	11.2
7/14/93	L1	0.5	0.2	24.7	7.1
	L1	2.0		24.6	7.0
	L1	3.5		22.0	0.9
	L2	0.5	0.1	24.4	6.0
	L3	0.5	0.2	24.4	6.8
7/28/93	L1	0.5	0.3	23.9	13.8
	L1	2.0		23.8	13.3
	L1	3.5		23.7	12.8
	L2	0.5	0.3	23.3	13.2

Table C2. Cont.

Date	Site	Depth (m)	Secchi (m)	Temp. (°C)	D.O. (mg/l)
7/28/93	L3	0.5	0.3	23.9	13.9
8/9/93	L1	0.5	0.3	22.8	13.7
	L1	2.0		22.6	12.5
	L1	3.5		22.2	12.2
	L2	0.5	0.3	23.1	14.0
	L3	0.5	0.3	23.0	14.4
8/30/93	L1	0.5	0.3	23.0	6.4
	L1	2.0		22.9	6.2
	L1	3.5		22.9	5.8
	L2	0.5	0.4	22.7	7.2
	L3	0.5	0.3	22.4	7.2
9/14/93	L1	0.5	0.3	17.4	9.1
	L1	2.0		17.3	9.0
	L1	3.5		17.2	9.3
	L2	0.5	0.4	17.0	9.4
	L3	0.5	0.4	16.9	10.4
9/26/93	L1	0.5	0.5	15.7	11.6
	L1	2.0		15.6	11.6
	L1	3.5		15.6	11.5
	L2	0.5	0.4	15.8	11.8
	L3	0.5	0.6	15.7	12.6
10/12/93	L1	0.5	0.8	11.4	10.1
	L1	2.0		11.4	10.0
	L1	3.5		11.3	9.9
	L2	0.5	0.1	11.1	10.1
	L3	0.5	0.9	11.2	10.5

Table C3. Measurements of pH, Total Alkalinity, and Chlorophyll a in 1993 at various sites in Swan Lake

Date	Site	Depth (m)	pH	Total Alkalinity (mg/L $CaCO_3$)	Chlorophyll a (mg/m^3)
4/27/93	L1	0.5	8.8		46.547
	L1	2.0	8.8		35.333
	L1	3.5	8.8		60.404
	L2	0.5	8.9		57.049
	L3	0.5	8.9		64.935
5/19/93	L1	0.5	8.6	148	57.079
	L1	2.0	8.6	147	49.595
	L1	3.5	8.6	148	56.204
	L2	0.5	8.6	148	57.446
	L3	0.5	8.7	148	16.139
5/26/93	L1	0.5	8.8	135	63.368
	L1	2.0	8.8	137	67.284
	L1	3.5	8.8	137	36.512
	L2	0.5	8.8	132	55.670
	L3	0.5	8.9	134	55.994
6/2/93	L1	0.5	9.0	119	22.962
	L1	2.0	9.0	122	60.342
	L1	3.5	9.1	124	109.559
	L2	0.5	9.1	123	124.651
	L3	0.5	9.0	128	146.469
6/9/93	L1	0.5	9.5	110	182.201
	L1	2.0	9.5	109	201.718
	L1	3.5	9.5	114	200.050
	L2	0.5	9.5	114	201.959
	L3	0.5	9.5	112	193.308
6/23/93	L1	0.5	9.5	108	140.086
	L1	2.0	9.5	112	133.678
	L1	3.5	9.2	115	82.058
	L2	0.5	9.7	109	203.810
	L3	0.5	9.8	109	266.146
7/14/93	L1	0.5	8.2	96	50.285
	L1	2.0	8.3	96	35.244
	L1	3.5	7.6	105	19.491
	L2	0.5	8.1	98	33.375
	L3	0.5	8.0	94	50.018

Table C3. Cont.

Date	Site	Depth (m)	pH	Total Alkalinity (mg/L $CaCO_3$)	Chlorophyll a (mg/m^3)
7/28/93	L1	0.5	8.8	122	115.745
	L1	2.0	8.8	121	92.115
	L1	3.5	8.8	122	96.254
	L2	0.5	8.7	129	84.906
	L3	0.5	8.8	125	67.658
8/9/93	L1	0.5	9.4	95	160.333
	L1	2.0	9.4	97	157.352
	L1	3.5	9.3	96	93.450
	L2	0.5	9.5	93	184.586
	L3	0.5	9.5	93	
8/30/93	L1	0.5	8.7	109	128.961
	L1	2.0	8.8	104	122.953
	L1	3.5	8.8	106	88.644
	L2	0.5	8.9	106	135.422
	L3	0.5	8.8	108	116.613
9/14/93	L1	0.5	8.6		99.458
	L1	2.0	8.7		92.316
	L1	3.5	8.7		129.940
	L2	0.5	8.7		93.984
	L3	0.5	8.8		126.425
9/26/93	L1	0.5	8.7	121	73.158
	L1	2.0	8.7	119	79.121
	L1	3.5	8.7	120	80.456
	L2	0.5	8.7	121	91.634
	L3	0.5	8.7	124	109.871
10/12/93	L1	0.5	8.5	140	34.225
	L1	2.0	8.5	133	36.045
	L1	3.5	8.6	136	36.686
	L2	0.5	8.6	135	29.157
	L3	0.5	8.6	135	33.108

Table C4. Measurements of total phosphorus, soluble reactive phosphorus, total suspended solids, inorganic suspended solids, and organic suspended solids at Swan Lake sites L1, L2, L3, and inlets, tiles, and wildlife pen sites in 1993

Date	Site	Depth (m)	Total Phos. (mg/L)	Soluble Reactive Phos. (mg/L)	Total Susp. Solids (mg/L)	Inorganic Susp. Solids (mg/L)	Organic Susp. Solids (mg/L)
3/10/93	INCW		0.656	0.449	3.24	1.38	1.86
	INCE						
	INSW		1.160	0.647	16.32	11.14	5.18
	INW		1.160	0.712	9.66	5.56	4.10
	INN1						
	INN2						
	INWP1						
	INSE		0.996	0.449	19.71	14.66	5.05
	INS		1.265	0.598	7.37	4.06	3.31
	INWP3		1.536	0.602	5.40	2.74	2.66
3/25/93	INCW		0.718	0.536	7.70	5.27	2.43
	INCE		0.718		27.33	20.42	6.91
	INSW		0.718	0.326	15.15	12.61	2.54
	INW		0.718	0.506	31.64	26.55	5.09
	INN1		0.718	0.747	6.06	2.13	3.93
	INN2		0.718	0.440	15.91	11.81	4.10
	INWP1		0.458	0.148	38.27	31.08	7.19
	INSE		0.718	0.747	28.50	23.53	4.97
	INS		0.718	0.747	16.93	13.10	3.83
	INWP3		0.718	0.747	77.95	67.15	10.80
3/31/93	INCW		0.718	0.277	98.45	84.15	14.30
	INCE		0.718	0.350	725.80	407.00	318.80
	INSW		0.139	0.056	1.43	1.20	0.23
	INW		0.718	0.109	65.47	50.07	15.40
	INN1		0.129	0.022	7.90	5.39	2.51
	INN2		0.183	0.056	12.24	9.96	2.28
	INWP1		0.718	0.010	169.52	146.08	23.44
	INSE		0.291	0.052	57.85	49.63	8.22
	INS		0.132	0.001	19.71	16.11	3.60
	INWP3		0.154	0.020	26.90	22.55	4.35
4/8/93	INCW		0.216	0.085	1.41	0.72	0.69
	INCE			0.114	15.74	8.40	7.34
	INSW		0.130	0.039	1.33	0.79	0.54
	INW		0.122	0.037	2.88	1.86	1.02
	INN1		0.076	0.015	1.59	1.24	0.35
	INN2		0.095	0.040	1.15	1.04	0.11
	INWP1						
	INSE		0.055	0.017	1.18	0.86	0.32
	INS		0.041	0.008	1.15	0.66	0.49

Table C4. Cont.

Date	Site	Depth (m)	Total Phos. (mg/L)	Soluble Reactive Phos. (mg/L)	Total Susp. Solids (mg/L)	Inorganic Susp. Solids (mg/L)	Organic Susp. Solids (mg/L)
4/8/93	INWP3		0.052	0.013	1.53	1.29	0.24
4/15/93	INCW		0.167	0.059	3.34	2.11	1.23
	INCE			0.872	20.67	9.69	10.98
	INSW		0.120	0.053	0.48	0.42	0.06
	INW		0.119	0.043	3.68	2.41	1.27
	INN1		0.050	0.011	3.51	2.83	0.68
	INN2		0.079	0.038	2.67	1.94	0.73
	INWP1		1.066	0.040	378.20	326.67	51.53
	INSE		0.102	0.043	6.80	5.63	1.17
	INS		0.044	0.006	3.73	2.28	1.45
	INWP3		0.048	0.011	4.15	3.28	0.87
4/27/93	L1	0.5	0.139	0.003	16.00	6.95	9.05
	L1	2.0	0.163	0.002	15.51	6.13	9.38
	L1	3.5	0.133	0.004	20.25	8.88	11.37
	L2	0.5	0.112	0.003	13.72	4.86	8.86
	L3	0.5	0.152	0.003	16.11	6.74	9.37
	INCW						
	INCE		0.057	0.029	2.87	1.95	0.92
	INSW		0.179	0.085	1.02	0.76	0.26
	INW		0.047	0.014	1.41	1.06	0.35
	INN1						
	INN2		0.066	0.025	8.34	7.11	1.23
	INWP1						
	INSE		0.035	0.014	0.96	0.64	0.32
	INS		0.029	0.012	0.48	0.34	0.14
	INWP3		0.036	0.009	2.39	1.65	0.74
5/19/93	L1	0.5	0.109	0.006	23.73	9.65	14.08
	L1	2.0	0.110	0.005	24.94	10.87	14.07
	L1	3.5	0.124	0.006	23.61	9.73	13.88
	L2	0.5	0.108	0.005	24.00	10.39	13.61
	L3	0.5	0.103	0.006	23.91	9.92	13.99
	INCW						
	INCE		0.064	0.048	1.94	1.21	0.73
	INSW		0.130	0.053	9.38	7.25	2.13
	INW		0.053	0.034	1.73	1.47	0.26
	INN1						
	INN2		0.077	0.050	4.16	3.25	0.91
	INWP1						
	INSE		0.036	0.027	2.19	1.80	0.39
	INS		0.026	0.024	0.34	0.21	0.13
	INWP3		0.041	0.027	3.00	2.09	0.91

Table C4. Cont.

Date	Site	Depth (m)	Total Phos. (mg/L)	Soluble Reactive Phos. (mg/L)	Total Susp. Solids (mg/L)	Inorganic Susp. Solids (mg/L)	Organic Susp. Solids (mg/L)
5/23/93	INCW						
	INCE		0.060	0.040	1.80	1.30	0.50
	INSW		0.111	0.051	1.05	0.53	0.52
	INW		0.062	0.033	3.22	2.14	1.08
	INN1		0.215	0.015	20.81	15.77	5.04
	INN2		0.074	0.047	5.62	4.72	0.90
	INWP1		0.219	0.012	14.82	9.68	5.14
	INSE		0.068	0.039	3.70	2.99	0.71
	INS		0.029	0.021	0.45	0.17	0.28
	INWP3		0.092	0.022	7.45	6.23	1.22
5/26/93	L1	0.5	0.173	0.003	29.61	10.26	19.35
	L1	2.0	0.142	0.004	26.46	11.61	14.85
	L1	3.5	0.163	0.004	23.43	8.64	14.79
	L2	0.5	0.150	0.004	28.66	9.11	19.55
	L3	0.5	0.156	0.004	28.66	10.01	18.65
	INCW						
	INCE		0.062	0.036	1.89	1.42	0.47
	INSW		0.108	0.040	0.87	0.44	0.43
	INW		0.092	0.036	3.22	2.44	0.78
	INN1						
	INN2		0.102	0.059	2.14	1.49	0.65
	INWP1						
	INSE		0.034	0.025	0.42	0.36	0.06
	INS		0.030	0.022	0.34	0.12	0.22
	INWP3		0.055	0.026	2.53	1.95	0.58
6/2/93	L1	0.5	0.167	0.000	37.27	11.73	25.54
	L1	2.0	0.145	0.000	37.09	10.31	26.78
	L1	3.5	0.136	0.000	37.94	11.69	26.25
	L2	0.5	0.151	0.001	44.72	12.64	32.08
	L3	0.5	0.154	0.000	42.76	16.55	26.21
	INCW						
	INCE		0.049	0.033	1.25	0.39	0.86
	INSW		0.103	0.050	1.06	0.54	0.52
	INW		0.093	0.046	5.01	4.12	0.89
	INN1						
	INN2		0.081	0.059	1.47	1.17	0.30
	INWP1		0.298	0.006	10.61	7.87	2.74
	INSE		0.057	0.039	0.89	0.51	0.38
	INS		0.035	0.019	1.12	0.55	0.57
	INWP3		0.073	0.025	3.01	2.98	0.03
6/9/93	L1	0.5	0.212	0.000	62.13	20.13	42.00

Table C4. Cont.

Date	Site	Depth (m)	Total Phos. (mg/L)	Soluble Reactive Phos. (mg/L)	Total Susp. Solids (mg/L)	Inorganic Susp. Solids (mg/L)	Organic Susp. Solids (mg/L)
6/9/93	L1	2.0	0.204	0.001	57.95	16.70	41.25
	L1	3.5	0.203	0.001	60.49	20.99	39.50
	L2	0.5	0.218	0.001	55.46	15.86	39.60
	L3	0.5	0.203	0.000	58.31	16.51	41.80
	INCW						
	INCE		0.044	0.023	1.01	0.22	0.79
	INSW		0.110	0.043	1.39	0.65	0.74
	INW		0.071	0.037	5.75	4.78	0.97
	INN1						
	INN2						
	INWP1						
	INSE		0.048	0.030	0.62	0.27	0.35
	INS		0.030	0.019	0.32	0.20	0.12
	INWP3		0.044	0.020	3.11	2.07	1.04
6/14/93	INCW		0.628	0.159	44.73	36.73	8.00
	INCE		0.073	0.027	3.42	2.16	1.26
	INSW			0.067	2.43	1.12	1.31
	INW		0.149	0.056	27.50	21.57	5.93
	INN1		0.229	0.014	21.16	17.53	3.63
	INN2		0.102	0.036	8.02	2.80	5.22
	INWP1						
	INSE		0.059	0.037	1.78	0.85	0.93
	INS		0.185	0.019	0.55	0.22	0.33
	INWP3		0.036	0.025	8.16	6.36	1.80
6/23/93	L1	0.5	0.211	0.003	47.00	15.92	31.08
	L1	2.0	0.227	0.003	46.60	15.60	31.00
	L1	3.5	0.212	0.008	39.56	14.76	24.80
	L2	0.5	0.283	0.002	65.88	15.84	50.04
	L3	0.5	0.319	0.002	77.11	15.44	61.67
	INCW						
	INCE		0.045	0.022	1.34	0.44	0.90
	INSW		0.115	0.060	1.14	0.56	0.58
	INW		0.072	0.039	2.68	2.10	0.58
	INN1						
	INN2						
	INWP1						
	INSE		0.051	0.031	1.50	1.23	0.27
	INS		0.033	0.019	0.21	0.10	0.11
	INWP3		0.045	0.019	2.37	1.65	0.72
6/30/93	INCW						
	INCE		0.055	0.031	1.30	0.64	0.66

Table C4. Cont.

Date	Site	Depth (m)	Total Phos. (mg/L)	Soluble Reactive Phos. (mg/L)	Total Susp. Solids (mg/L)	Inorganic Susp. Solids (mg/L)	Organic Susp. Solids (mg/L)
6/30/93	INSW		0.190	0.114	2.07	1.34	0.73
	INW		0.140	0.065			
	INN1						
	INN2		0.105	0.060	1.43	1.01	0.42
	INWP1		0.447	0.045	19.03	14.01	5.02
	INSE		0.079	0.031	1.80	1.49	0.31
	INS		0.044	0.021	0.67	0.22	0.45
	INWP3		0.072	0.031	3.63	2.66	0.97
7/1/93	INCW						
	INCE		0.063	0.032	1.49	0.78	0.71
	INSW		0.187	0.115	1.94	1.16	0.78
	INW		0.117	0.054	5.34	4.39	0.95
	INN1						
	INN2		0.130	0.056	7.17	5.82	1.35
	INWP1		0.459	0.014	22.95	14.47	8.48
	INSE		0.071	0.030	1.84	1.33	0.51
	INS		0.045	0.016	3.93	2.27	1.66
	INWP3		0.078	0.028	11.03	4.68	6.35
7/9/93	INCW		1.852	0.517	561.80	487.90	73.90
	INCE						
	INSW		1.485	0.636	223.53	193.60	29.93
	INW		1.895	0.418	433.90	373.90	60.00
	INN1		0.885	0.067	23.42	19.78	3.64
	INN2						
	INWP1		1.025	0.137	243.50	204.70	38.80
	INSE						
	INS						
	INWP3		1.235	0.333	204.90	175.90	29.00
7/14/93	L1	0.5	0.438	0.091	44.34	26.61	17.73
	L1	2.0	0.420	0.088	47.80	28.43	19.37
	L1	3.5	0.381	0.105	42.20	29.43	12.77
	L2	0.5	0.393	0.090	44.33	27.20	17.13
	L3	0.5	0.408	0.089	43.60	26.83	16.77
	INCW						
	INCE		0.091	0.044	1.39	0.99	0.40
	INSW		0.158	0.076	6.69	5.58	1.11
	INW		0.126	0.042	4.77	3.85	0.92
	INN1		0.645	0.031	35.12	29.54	5.58
	INN2		0.107	0.048	4.61	3.64	0.97
	INWP1						
	INSE		0.059	0.028	3.42	2.76	0.66

Table C4. Cont.

Date	Site	Depth (m)	Total Phos. (mg/L)	Soluble Reactive Phos. (mg/L)	Total Susp. Solids (mg/L)	Inorganic Susp. Solids (mg/L)	Organic Susp. Solids (mg/L)
7/14/93	INS		0.041	0.021	1.77	1.43	0.34
	INWP3		0.061	0.023	3.10	2.39	0.71
7/28/93	L1	0.5	0.189	0.020	32.08	16.82	15.26
	L1	2.0	0.154	0.002	32.76	17.46	15.30
	L1	3.5	0.181	0.007	38.23	22.53	15.70
	L2	0.5	0.187	0.002	38.93	23.08	15.85
	L3	0.5	0.150	0.003	28.32	15.48	12.84
	INCW						
	INCE		0.053	0.026	1.59	0.74	0.85
	INSW		0.122	0.057	4.93	3.80	1.13
	INW		0.055	0.025	1.28	0.62	0.66
	INN1		0.189	0.016	31.70	27.00	4.70
	INN2		0.075	0.038	3.74	2.80	0.94
	INWP1		1.125	0.001	21.60	15.10	6.50
	INSE		0.050	0.026	0.83	0.27	0.56
	INS		0.030	0.013	0.72	0.28	0.44
	INWP3		0.036	0.018	1.86	1.33	0.53
8/9/93	L1	0.5	0.184	0.002	54.40	19.60	34.80
	L1	2.0	0.176	0.004	56.10	24.10	32.00
	L1	3.5	0.156	0.003	42.47	21.57	20.90
	L2	0.5	0.160	0.003	51.55	17.70	33.85
	L3	0.5	0.173	0.004	54.00	19.60	34.40
	INCW						
	INCE		0.035	0.023	2.22	1.14	1.08
	INSW		0.104	0.061	9.83	7.01	2.82
	INW		0.036	0.027	1.80	1.16	0.64
	INN1		0.062	0.008	21.48	17.40	4.08
	INN2		0.064	0.050	1.48	0.73	0.75
	INWP1						
	INSE		0.027	0.023	1.69	1.24	0.45
	INS		0.015	0.014	0.76	0.17	0.59
	INWP3		0.024	0.019	3.05	1.39	1.66
8/18/93	INCW						
	INCE		0.185	0.101	10.52	7.88	2.64
	INSW		0.299	0.270	8.18	5.79	2.39
	INW			0.209	17.03	14.11	2.92
	INN1		0.190	0.040	108.08	93.12	14.96
	INN2		0.138	0.071	4.96	3.90	1.06
	INWP1						
	INSE		0.077	0.038	4.75	3.03	1.72
	INS		0.051	0.022	2.75	1.67	1.08

Table C4. Cont.

Date	Site	Depth (m)	Total Phos. (mg/L)	Soluble Reactive Phos. (mg/L)	Total Susp. Solids (mg/L)	Inorganic Susp. Solids (mg/L)	Organic Susp. Solids (mg/L)
8/18/93	INWP3		0.108	0.037	11.32	9.14	2.18
8/29/93	INCW		1.228	1.068	54.09	43.02	11.07
	INCE		0.212	0.097	6.11	4.73	1.38
	INSW		0.350	0.247	11.25	7.58	3.67
	INW		0.236	0.140	11.10	8.50	2.60
	INN1						
	INN2		0.148	0.069	4.14	2.87	1.27
	INWP1		0.368	0.108	29.08	21.88	7.20
	INSE		0.107	0.040	4.14	3.21	0.93
	INS		0.059	0.027	1.76	1.16	0.60
	INWP3		0.121	0.040	11.74	9.15	2.59
8/30/93	L1	0.5	0.161	0.004	26.48	12.95	13.53
	L1	2.0	0.146	0.004	23.67	10.85	12.82
	L1	3.5	0.144	0.004	26.98	13.05	13.93
	L2	0.5	0.139	0.004	26.47	11.78	14.69
	L3	0.5	0.177	0.004	29.55	14.70	14.85
	INCW						
	INCE		2.200	1.035	485.73	327.27	158.46
	INSW		0.270	0.194	36.79	29.64	7.15
	INW		0.464	0.263	37.81	27.66	10.15
	INN1		0.168	0.045	29.48	24.21	5.27
	INN2		0.170	0.086	6.86	4.93	1.93
	INWP1		0.311	0.045	27.68	20.82	6.86
	INSE		0.309	0.142	43.15	35.18	7.97
	INS		0.060	0.026	2.78	1.79	0.99
	INWP3		0.231	0.151	9.82	7.80	2.02
9/14/93	L1	0.5	0.233	0.003	46.03	27.60	18.43
	L1	2.0	0.239	0.004	48.10	27.47	20.63
	L1	3.5	0.223	0.004	44.37	24.87	19.50
	L2	0.5	0.222	0.006	45.47	24.23	21.24
	L3	0.5	0.188	0.004	34.53	15.85	18.68
	INCW						
	INCE		0.042	0.019	1.47	0.75	0.72
	INSW		0.103	0.052	17.22	12.09	5.13
	INW		0.063	0.032	1.69	1.12	0.57
	INN1						
	INN2		0.092	0.051	0.59	0.28	0.31
	INWP1						
	INSE		0.044	0.026	0.76	0.36	0.40
	INS		0.033	0.018	0.32		0.32
	INWP3		0.050	0.019	1.05	0.74	0.31

Table C4. Cont.

Date	Site	Depth (m)	Total Phos. (mg/L)	Soluble Reactive Phos. (mg/L)	Total Susp. Solids (mg/L)	Inorganic Susp. Solids (mg/L)	Organic Susp. Solids (mg/L)
9/26/93	L1	0.5	0.136	0.003	21.25	6.60	14.65
	L1	2.0	0.122	0.003	22.42	6.64	15.78
	L1	3.5	0.138	0.003	25.20	7.73	17.47
	L2	0.5	0.110	0.005	21.09	6.78	14.31
	L3	0.5	0.137	0.004	23.36	4.29	19.07
	INCW						
	INCE		0.043	0.017	3.34	2.67	0.67
	INSW		0.105	0.054	38.53	28.47	10.06
	INW		0.067	0.034	1.58	1.04	0.54
	INN1						
	INN2		0.106	0.061	1.89	0.27	1.62
	INWP1						
	INSE		0.076	0.038	5.41	4.49	0.92
	INS		0.037	0.018	15.66	10.79	4.87
	INWP3		0.045	0.018	2.50	0.89	1.61
10/12/93	L1	0.5	0.068	0.005	11.68	4.60	7.08
	L1	2.0	0.075	0.004	12.59	4.46	8.13
	L1	3.5	0.067	0.004	10.75	4.49	6.26
	L2	0.5	0.083	0.005	10.36	3.19	7.17
	L3	0.5	0.070	0.004	11.94	4.35	7.59
	INCW						
	INCE		0.043	0.019	10.46	8.07	2.39
	INSW		0.101	0.045	1.83	0.13	1.70
	INW		0.061	0.030	0.75	0.36	0.39
	INN1						
	INN2		0.113	0.075	1.38	0.57	0.81
	INWP1						
	INSE		0.046	0.029	4.42	2.30	2.12
	INS		0.035	0.019	0.53	0.23	0.30
	INWP3		0.053	0.018	17.65	12.26	5.39

Table C5. Measurements of total nitrogen, nitrate nitrogen, ammonia nitrogen, and organic nitrogen at Swan Lake sites L1, L2, L3, and inlets, tiles and wildlife pen sites in 1993

Date	Site	Depth (m)	Total Nitrogen (mg/L)	Nitrate Nitrogen (mg/L)	Ammonia Nitrogen (mg/L)	Organic Nitrogen (mg/L)
3/10/93	INCW		2.30	0.95		
	INCE					
	INSW		7.04	5.50		
	INW		6.84	5.34		
	INN1					
	INN2					
	INWP1					
	INSE		5.58	4.20		
	INS		7.18	2.01		
	INWP3		7.12	4.40		
3/25/93	INCW		2.88	1.35	0.58	0.95
	INCE		12.49	4.97	7.09	0.43
	INSW		8.76	7.72	0.62	0.42
	INW		4.74	3.51	1.21	0.02
	INN1		3.53	0.67	1.67	1.19
	INN2		3.12	1.63	1.05	0.44
	INWP1		1.56	0.28	0.72	0.56
	INSE		2.10	0.43	0.99	0.68
	INS		4.70	1.58	2.77	0.35
	INWP3		5.84	1.92	3.39	0.53
3/31/93	INCW		1.65	0.75	0.19	0.71
	INCE					
	INSW		20.96	17.20	0.15	3.61
	INW		4.74	3.50	0.88	0.36
	INN1		6.40	5.99	0.21	0.20
	INN2		7.12	6.18	0.20	0.74
	INWP1		1.39	0.22	0.27	0.90
	INSE		11.12	8.12	0.09	2.91
	INS		11.68	8.84	0.06	2.78
	INWP3		12.12	10.08	0.16	1.88
4/8/93	INCW		1.04	0.38	0.14	0.52
	INCE		1.85	0.87	0.12	0.86
	INSW		32.21	29.68	0.00	2.53
	INW		16.77	15.85	0.00	0.92
	INN1		13.12	11.72	0.00	1.40
	INN2		7.86	7.80	0.00	0.06
	INWP1					
	INSE		6.22	3.71	0.00	2.51
	INS		6.90	3.86	0.00	3.04
	INWP3		15.43	14.44	0.00	0.99

Table C5. Cont.

Date	Site	Depth (m)	Total Nitrogen (mg/L)	Nitrate Nitrogen (mg/L)	Ammonia Nitrogen (mg/L)	Organic Nitrogen (mg/L)
4/15/93	INCW		0.80	0.21	0.00	0.59
	INCE		1.87	1.53	0.31	0.03
	INSW		31.56	30.08	0.00	1.48
	INW		20.77	18.69	0.14	1.94
	INN1		0.25	0.06	0.00	0.19
	INN2		7.92	7.44	0.00	0.48
	INWP1		0.61	0.05	0.26	0.30
	INSE		7.87	3.75	0.00	4.12
	INS		7.77	4.65	0.00	3.12
	INWP3		6.23	1.60	0.00	4.63
4/27/93	L1	0.5	3.37	1.71	0.03	1.63
	L1	2.0	3.71	1.72	0.03	1.96
	L1	3.5	3.38	1.72	0.03	1.63
	L2	0.5	3.38	1.74	0.02	1.62
	L3	0.5	3.68	1.78	0.02	1.88
	INCW					
	INCE		1.46	1.03	0.04	0.39
	INSW		33.47	28.65	0.03	4.79
	INW		22.20		0.02	
	INN1					
	INN2		5.91	5.85	0.03	0.03
	INWP1					
	INSE		3.32	0.07	0.02	3.23
	INS			22.43	0.01	
	INWP3		13.88	13.36	0.03	0.49
5/19/93	L1	0.5	3.04	1.86	0.00	1.18
	L1	2.0	2.98	1.84	0.00	1.14
	L1	3.5	2.91	1.87	0.00	1.04
	L2	0.5	3.29	1.90	0.00	1.39
	L3	0.5	3.21	1.96	0.00	1.25
	INCW					
	INCE		1.36	0.80	0.00	0.56
	INSW		28.32	27.18	0.00	1.14
	INW		29.23	29.11	0.00	0.12
	INN1					
	INN2		4.91	4.79	0.00	0.12
	INWP1					
	INSE		3.45	2.49	0.00	0.96
	INS		10.56	9.05	0.00	1.51
	INWP3		20.84	18.30	0.00	2.54
5/23/93	INCW					
	INCE		1.15	0.69	0.03	0.43

Table C5. Cont.

Date	Site	Depth (m)	Total Nitrogen (mg/L)	Nitrate Nitrogen (mg/L)	Ammonia Nitrogen (mg/L)	Organic Nitrogen (mg/L)
5/23/93	INSW		33.76	27.69	0.02	6.05
	INW		25.94	25.41	0.03	0.50
	INN1		4.68	4.58	0.06	0.04
	INN2		5.03	4.87	0.02	0.14
	INWP1		0.60	0.11	0.02	0.47
	INSE		3.91	3.21	0.14	0.56
	INS		6.60	5.41	0.00	1.19
	INWP3		3.25	2.38	0.02	0.85
5/26/93	L1	0.5	3.76	1.69	0.04	2.03
	L1	2.0	3.77	1.68	0.03	2.06
	L1	3.5	3.66	1.64	0.03	1.99
	L2	0.5	3.75	1.82	0.03	1.90
	L3	0.5	3.68	1.71	0.03	1.94
	INCW					
	INCE		1.34	1.02	0.03	0.29
	INSW		34.17	30.76	0.02	3.39
	INW		23.88	20.40	0.03	3.45
	INN1					
	INN2		4.20	4.04	0.01	0.15
	INWP1					
	INSE		3.94	2.11	0.02	1.81
	INS		0.82	0.53	0.00	0.29
	INWP3		21.52	20.30	0.02	1.20
6/2/93	L1	0.5	3.48	2.15	0.00	1.33
	L1	2.0	3.98	1.26	0.00	2.72
	L1	3.5	3.53	1.24	0.00	2.29
	L2	0.5	4.38	1.26	0.00	3.12
	L3	0.5	4.16	1.28	0.00	2.88
	INCW					
	INCE		18.16	12.32	0.00	5.84
	INSW		36.72	29.72	0.00	7.00
	INW		21.60	20.12	0.01	1.47
	INN1					
	INN2		4.03	4.01	0.00	0.02
	INWP1		0.56	0.30	0.00	0.26
	INSE		16.32	13.56	0.00	2.76
	INS		17.52	16.20	0.00	1.32
	INWP3		18.92	17.04	0.00	1.88
6/9/93	L1	0.5	3.88	0.79	0.01	3.08
	L1	2.0	4.24	0.84	0.01	3.39
	L1	3.5	3.94	0.84	0.01	3.09
	L2	0.5	4.23	0.89	0.00	3.34

Table C5. Cont.

Date	Site	Depth (m)	Total Nitrogen (mg/L)	Nitrate Nitrogen (mg/L)	Ammonia Nitrogen (mg/L)	Organic Nitrogen (mg/L)
6/9/93	L3	0.5	3.85	0.87	0.00	2.98
	INCW					
	INCE			11.52	0.00	
	INSW		32.35	31.10	0.00	1.25
	INW		22.65	21.10	0.04	1.51
	INN1					
	INN2					
	INWP1					
	INSE		7.52	6.26	0.08	1.18
	INS		18.32	16.00	0.00	2.32
	INWP3		22.48	22.00	0.00	0.48
6/14/93	INCW		2.53	2.02	0.08	0.43
	INCE		31.92	29.92	0.00	2.00
	INSW		34.45	29.52	0.00	4.93
	INW		25.36	22.72	0.06	2.58
	INN1		1.74	0.91	0.22	0.61
	INN2		3.86	3.67	0.03	0.16
	INWP1					
	INSE		17.44	15.84	0.07	1.53
	INS		19.10	18.20	0.00	0.90
	INWP3		19.80	19.68	0.02	0.10
6/23/93	L1	0.5	2.97	0.61	0.13	2.23
	L1	2.0	3.76	0.63	0.15	2.98
	L1	3.5	2.75	0.59	0.22	1.94
	L2	0.5	4.71	0.59	0.03	4.09
	L3	0.5	4.30	0.59	0.00	3.71
	INCW					
	INCE		1.81	1.44	0.00	0.37
	INSW		34.27	30.60	0.00	3.67
	INW		27.40	23.12	0.00	4.28
	INN1					
	INN2					
	INWP1					
	INSE			16.14	0.07	
	INS		18.88	17.05	0.00	1.83
	INWP3		26.24	23.00	0.00	3.24
6/30/93	INCW					
	INCE		1.32	0.44	0.00	0.88
	INSW		29.20	28.30	0.00	0.90
	INW		22.50	20.40	0.02	2.08
	INN1					
	INN2		3.01	2.74	0.00	0.27

Table C5. Cont.

Date	Site	Depth (m)	Total Nitrogen (mg/L)	Nitrate Nitrogen (mg/L)	Ammonia Nitrogen (mg/L)	Organic Nitrogen (mg/L)
6/30/93	INWP1		1.00	0.06	0.13	0.81
	INSE		5.95	3.58	0.05	2.32
	INS		18.55	16.70	0.00	1.85
	INWP3		18.90	16.90	0.01	1.99
7/1/93	INCW					
	INCE		39.95	36.36	0.02	3.57
	INSW		29.55	27.65	0.00	1.90
	INW		22.40	21.10	0.01	1.29
	INN1					
	INN2		3.32	2.30	0.00	1.02
	INWP1		0.75	0.00	0.02	0.73
	INSE		13.25	2.94	0.00	10.31
	INS		18.00	17.85	0.00	0.15
	INWP3		16.98	15.50	0.00	1.48
7/9/93	INCW		12.90	10.86	0.29	1.75
	INCE					
	INSW		16.48	16.20	0.25	0.03
	INW		17.50	13.68	0.25	3.57
	INN1		7.65	7.20	0.00	0.45
	INN2					
	INWP1		2.01	0.86	0.00	1.15
	INSE					
	INS					
	INWP3		5.02	4.06	0.00	0.96
7/14/93	L1	0.5	4.66	2.23	0.40	2.03
	L1	2.0	4.60	2.13	0.44	2.03
	L1	3.5	4.89	3.27	0.77	0.85
	L2	0.5	4.26	2.25	0.55	1.46
	L3	0.5	4.19	2.19	0.57	1.43
	INCW					
	INCE		35.00	33.70	0.00	1.30
	INSW		34.57	32.88	0.00	1.69
	INW		25.96	23.18	0.00	2.78
	INN1		11.04	10.36	0.00	0.68
	INN2		4.85	4.66	0.00	0.19
	INWP1					
	INSE		17.36	16.02	0.02	1.32
	INS		23.22	20.80	0.00	2.42
	INWP3		17.58	15.60	0.00	1.98
7/28/93	L1	0.5	4.79	3.19	0.17	1.43
	L1	2.0	5.04	3.20	0.17	1.67

Table C5. Cont.

Date	Site	Depth (m)	Total Nitrogen (mg/L)	Nitrate Nitrogen (mg/L)	Ammonia Nitrogen (mg/L)	Organic Nitrogen (mg/L)
7/28/93	L1	3.5	4.72	3.27	0.19	1.26
	L2	0.5	4.88	3.29	0.24	1.35
	L3	0.5	4.61	3.10	0.21	1.30
	INCW					
	INCE		10.87	7.80	0.06	3.01
	INSW		31.63	29.72	0.04	1.87
	INW		23.94	23.28	0.02	0.64
	INN1		8.07	7.38	0.08	0.61
	INN2		2.88	2.72	0.03	0.13
	INWP1		0.10	0.02	0.06	0.02
	INSE		16.17	16.08	0.08	0.01
	INS		17.42	17.02	0.03	0.37
	INWP3		20.22	19.34	0.02	0.86
8/9/93	L1	0.5	4.02	2.05	0.03	1.94
	L1	2.0	4.10	1.97	0.07	2.06
	L1	3.5	3.84	2.14	0.07	1.63
	L2	0.5	3.87	2.05	0.08	1.74
	L3	0.5	3.92	1.99	0.08	1.85
	INCW					
	INCE		13.01	9.22	0.06	3.73
	INSW		29.82	28.49	0.09	1.24
	INW		22.42	22.33	0.07	0.02
	INN1		3.44	2.85	0.53	0.06
	INN2		2.10	1.84	0.11	0.15
	INWP1					
	INSE		15.76	15.71	0.04	0.01
	INS		19.11	18.69	0.06	0.36
	INWP3		19.66	19.46	0.05	0.15
8/18/93	INCW					
	INCE		30.54	26.36	0.17	4.01
	INSW		22.43	21.08	0.12	1.23
	INW		14.99	14.19	0.11	0.69
	INN1		4.44	4.14	0.25	0.05
	INN2		2.49	2.37	0.10	0.02
	INWP1					
	INSE		14.42	13.84	0.13	0.45
	INS		16.53	16.27	0.07	0.19
	INWP3		15.04	14.13	0.08	0.83
8/29/93	INCW		4.25	1.83	0.11	2.31
	INCE		21.06	20.88	0.05	0.13
	INSW		22.62	21.19	0.13	1.30
	INW		16.11	14.69	0.10	1.32

Table C5. Cont.

Date	Site	Depth (m)	Total Nitrogen (mg/L)	Nitrate Nitrogen (mg/L)	Ammonia Nitrogen (mg/L)	Organic Nitrogen (mg/L)
8/29/93	INN1					
	INN2		2.89	2.44	0.14	0.31
	INWP1		0.43	0.05	0.17	0.21
	INSE		13.40	12.84	0.11	0.45
	INS		15.84	15.21	0.15	0.48
	INWP3		13.44	13.30	0.14	0.00
8/30/93	L1	0.5	3.06	0.78	0.90	1.38
	L1	2.0	3.05	0.78	0.83	1.44
	L1	3.5	2.94	0.77	1.07	1.10
	L2	0.5	2.88	0.73	0.57	1.58
	L3	0.5	3.21	0.94	0.56	1.71
	INCW					
	INCE		17.30	13.46		
	INSW		25.72	25.48	0.21	0.03
	INW		16.75	15.90	0.75	0.10
	INN1		7.94	7.32	0.37	0.25
	INN2		3.89	3.29	0.26	0.34
	INWP1		0.29	0.03	0.24	0.02
	INSE		10.84	10.72	0.11	0.01
	INS		15.00	14.90	0.06	0.04
	INWP3		15.61	14.50	0.24	0.87
9/14/93	L1	0.5	3.45	1.33	0.35	1.77
	L1	2.0	3.72	1.31	0.34	2.07
	L1	3.5	3.72	1.31	0.31	2.10
	L2	0.5	3.66	1.33	0.33	2.00
	L3	0.5	3.52	1.31	0.29	1.92
	INCW					
	INCE		13.36	9.06	0.00	4.30
	INSW		30.23	28.92	0.00	1.31
	INW		21.13	21.10	0.00	0.03
	INN1					
	INN2		2.25	1.92	0.00	0.33
	INWP1					
	INSE		14.35	14.28	0.00	0.07
	INS		15.99	15.47	0.00	0.52
	INWP3		17.42	16.92	0.00	0.50
9/26/93	L1	0.5	2.80	1.29	0.04	1.47
	L1	2.0	3.11	1.24	0.04	1.83
	L1	3.5	2.95	1.26	0.05	1.64
	L2	0.5	2.99	1.30	0.06	1.63
	L3	0.5	3.41	1.58	0.04	1.79
	INCW					

Table C5. Cont.

Date	Site	Depth (m)	Total Nitrogen (mg/L)	Nitrate Nitrogen (mg/L)	Ammonia Nitrogen (mg/L)	Organic Nitrogen (mg/L)
9/26/93	INCE		13.71	9.29	0.00	4.42
	INSW		28.15	27.65	0.01	0.49
	INW		20.44	19.72	0.00	0.72
	INN1					
	INN2		1.19	0.97	0.00	0.22
	INWP1					
	INSE		13.97	13.82	0.04	0.11
	INS		13.50	13.12	0.00	0.38
	INWP3		14.82	14.33	0.01	0.48
10/12/93	L1	0.5	2.49	1.18	0.08	1.23
	L1	2.0	2.93	1.13	0.10	1.70
	L1	3.5	2.44	1.14	0.09	1.21
	L2	0.5	2.41	1.12	0.10	1.19
	L3	0.5	2.45	1.19	0.08	1.18
	INCW					
	INCE		11.79	7.83	0.00	3.96
	INSW		29.47	27.67	0.00	1.80
	INW		18.83	18.74	0.00	0.09
	INN1					
	INN2		0.07	0.03	0.00	0.04
	INWP1					
	INSE		14.33	14.32	0.00	0.01
	INS		13.51	13.47	0.00	0.04
	INWP3		16.03	15.82	0.01	0.20

Table C6. Algae genera and biovolumes from site L1 for the upper mixing zone in Swan Lake in 1993

Date	Class	Genus	Cells/ml	Cell Volume per ml
5/19/93	Chlorophyta	*Closterium*	0	0
		Coelastrum	0	0
		Micrasterias	0	0
		Pediastrum	128	1382270
		Scenedesmus	128	220663
		Ulothrix	0	0
	Cyanophyta	*Anacystis*	98852	11170281
		Aphanizomenon	0	0
		Oscillatoria	0	0
	Bacillariophyceae	*Asterionella*	893	313393
		Navicula	0	0
		Stephanodiscus	574	17735969
	Crypyophyta	*Cryptomonas*	702	454592

Date	Class	Genus	Cells/ml	Cell Volume per ml
5/26/93	Chlorophyta	*Closterium*	64	15306
		Coelastrum	64	163903
		Micrasterias	0	0
		Pediastrum	0	0
		Scenedesmus	446	772321
		Ulothrix	0	0
	Cyanophyta	*Anacystis*	65944	7451658
		Aphanizomenon	0	0
		Oscillatoria	0	0
	Bacillariophyceae	*Asterionella*	5995	2104209
		Navicula	64	275510
		Stephanodiscus	510	15765306
	Crypyophyta	*Cryptomonas*	128	82653

Table C6. Cont.

Date	Class	Genus	Cells/ml	Cell Volume per ml
6/2/93	Chlorophyta	*Closterium*	191	45918
		Coelastrum	0	0
		Micrasterias	0	0
		Pediastrum	0	0
		Scenedesmus	0	0
		Ulothrix	0	0
	Cyanophyta	*Anacystis*	179847	20322704
		Aphanizomenon	0	0
		Oscillatoria	0	0
	Bacillariophyceae	*Asterionella*	2296	805867
		Navicula	0	0
		Stephanodiscus	255	7882653
	Crypyophyta	*Cryptomonas*	64	41327

Date	Class	Genus	Cells/ml	Cell Volume per ml
6/9/93	Chlorophyta	*Closterium*	128	30612
		Coelastrum	0	0
		Micrasterias	0	0
		Pediastrum	0	0
		Scenedesmus	0	0
		Ulothrix	0	0
	Cyanophyta	*Anacystis*	164860	18629145
		Aphanizomenon	0	0
		Oscillatoria	1594	1532207
	Bacillariophyceae	*Asterionella*	1913	671556
		Navicula	0	0
		Stephanodiscus	191	5911990
	Crypyophyta	*Cryptomonas*	0	0

Table C6. Cont.

Date	Class	Genus	Cells/ml	Cell Volume per ml
6/23/93	Chlorophyta	*Closterium*	0	0
		Coelastrum	0	0
		Micrasterias	0	0
		Pediastrum	0	0
		Scenedesmus	0	0
		Ulothrix	0	0
	Cyanophyta	*Anacystis*	88010	9945153
		Aphanizomenon	0	0
		Oscillatoria	0	0
	Bacillariophyceae	*Asterionella*	0	0
		Navicula	0	0
		Stephanodiscus	64	1970663
	Crypyophyta	*Cryptomonas*	0	0

Date	Class	Genus	Cells/ml	Cell Volume per ml
7/14/93	Chlorophyta	*Closterium*	0	0
		Coelastrum	0	0
		Micrasterias	0	0
		Pediastrum	0	0
		Scenedesmus	0	0
		Ulothrix	0	0
	Cyanophyta	*Anacystis*	26467	2990753
		Aphanizomenon	0	0
		Oscillatoria	0	0
	Bacillariophyceae	*Asterionella*	0	0
		Navicula	0	0
		Stephanodiscus	0	0
	Crypyophyta	*Cryptomonas*	0	0

Table C6. Cont.

Date	Class	Genus	Cells/ml	Cell Volume per ml
7/28/93	Chlorophyta	*Closterium*	0	0
		Coelastrum	0	0
		Micrasterias	0	0
		Pediastrum	0	0
		Scenedesmus	0	0
		Ulothrix	0	0
	Cyanophyta	*Anacystis*	36033	4071747
		Aphanizomenon	0	0
		Oscillatoria	0	0
	Bacillariophyceae	*Asterionella*	0	0
		Navicula	64	275510
		Stephanodiscus	1467	45325255
	Crypyophyta	*Cryptomonas*	128	82653

Date	Class	Genus	Cells/ml	Cell Volume per ml
8/9/93	Chlorophyta	*Closterium*	128	30612
		Coelastrum	0	0
		Micrasterias	0	0
		Pediastrum	0	0
		Scenedesmus	0	0
		Ulothrix	0	0
	Cyanophyta	*Anacystis*	182398	20610969
		Aphanizomenon	2041	9816327
		Oscillatoria	0	0
	Bacillariophyceae	*Asterionella*	0	0
		Navicula	0	0
		Stephanodiscus	255	7882653
	Crypyophyta	*Cryptomonas*	64	41327

Table C6. Cont.

Date	Class	Genus	Cells/ml	Cell Volume per ml
8/30/93	Chlorophyta	*Closterium*	0	0
		Coelastrum	0	0
		Micrasterias	0	0
		Pediastrum	0	0
		Scenedesmus	0	0
		Ulothrix	8355	10192602
	Cyanophyta	*Anacystis*	11480	1297194
		Aphanizomenon	2487	11963648
		Oscillatoria	0	0
	Bacillariophyceae	*Asterionella*	0	0
		Navicula	0	0
		Stephanodiscus	574	17735969
	Crypyophyta	*Cryptomonas*	255	165306

Date	Class	Genus	Cells/ml	Cell Volume per ml
9/14/93	Chlorophyta	*Closterium*	0	0
		Coelastrum	0	0
		Micrasterias	0	0
		Pediastrum	0	0
		Scenedesmus	0	0
		Ulothrix	0	0
	Cyanophyta	*Anacystis*	54209	6125638
		Aphanizomenon	0	0
		Oscillatoria	3508	3370855
	Bacillariophyceae	*Asterionella*	0	0
		Navicula	0	0
		Stephanodiscus	383	11823980
	Crypyophyta	*Cryptomonas*	64	41327

Table C6. Cont.

Date	Class	Genus	Cells/ml	Cell Volume per ml
9/26/93	Chlorophyta	*Closterium*	0	0
		Coelastrum	0	0
		Micrasterias	64	442602
		Pediastrum	0	0
		Scenedesmus	0	0
		Ulothrix	0	0
	Cyanophyta	*Anacystis*	216837	24502551
		Aphanizomenon	0	0
		Oscillatoria	0	0
	Bacillariophyceae	*Asterionella*	0	0
		Navicula	0	0
		Stephanodiscus	191	5911990
	Crypyophyta	*Cryptomonas*	191	123980