U.S. Army
Corps of Engineers

City of Attleboro
Town of North Attleboro

Lake Como Restoration Study

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EXECUTIVE SUMMARY

Lake Como is a 7-acre waterbody located in the City of Attleboro and the Town of North Attleboro, Massachusetts that experiences eutrophic conditions resulting in nuisance aquatic vegetation during the summertime. The Lake Como Restoration Study was undertaken by Massachusetts Department of Environmental Management, the City of Attleboro, and the Town of North Attleboro with assistance from the US Army Corps of Engineers. The study included an investigation and characterization of existing conditions and identification of alternatives to restore the Lake. Brief summaries of the Lake Como field program, watershed modeling evaluation, and restoration recommendations are provided below.

Lake Como Field Program

A series of field surveys were conducted from September 2000 through June 2002 to characterize present conditions in Lake Como in terms of physical, hydrologic, water quality, sediment quality, and biological characteristics. All observations obtained during the field program are consistent with characterization of Lake Como as a small, highly eutrophic pond system. Lake Como was observed to receive excessive nutrient loading combined with insufficient water volume to maintain healthy water quality conditions. The field program resulted in several key observations of present conditions in the Lake Como system including the following:

- <u>Leak in Outlet Dam</u> – A continuous leak was identified in the main pond's outlet dam effectively reducing the water level and storage capacity in the Lake Como system.
- <u>Low In-Lake Dissolved Oxygen Concentrations</u> – D.O. concentrations in the two ponds were frequently below the water quality standard of 5.0 mg/L, with measurements as low as 1.0 mg/L obtained. Low D.O. concentrations are due to the presence of an excess of aquatic vegetation in the ponds.
- <u>Excessive In-Lake Nutrient Concentrations</u> - Phosphorus and nitrogen concentrations in the two ponds were observed to be excessive during field surveys and were sufficient to support eutrophic conditions. For example, in-lake phosphorus concentrations ranged from 30 to 310 µg/L, with a mean value of 90 µg/L. Target in-lake phosphorus concentrations of 16 µg/L to 31 µg/Lwere established for Lake Como based on widely accepted evaluation guidelines. Thus, ambient phosphorus concentrations were observed to be approximately 3 to 5 times higher than acceptable levels indicating that nutrient levels must be dramatically reduced before water quality improvements may be achieved.
- <u>Excessive Stormwater Nutrient Concentrations</u> – Phosphorus and nitrogen concentrations from the 4 storm drains were observed to be excessive during wet-weather surveys and appeared to be a major source of elevated in-lake nutrient concentrations. For example,

phosphorus stormwater concentrations ranged from 60 to 6,200 μg/L, with a mean value of 1,760 μg/L.

- Sediment Nutrients – Nutrient levels in sediments were moderate to high indicating that sediments may act as a significant source in the overall nutrient budget, depending on oxygen levels, pH, and other factors.
- Extensive Aquatic Biological Growth - Extensive growth of phytoplankton and rooted aquatic vegetation was observed during the summertime survey. The water surface was covered with floating macrophytes diminishing the potential for recreational uses.

Lake Como Watershed Modeling Evaluation

A screening level watershed modeling evaluation was conducted on the Lake Como system to identify sources of impairment and to support evaluation lake restoration alternatives. The watershed modeling evaluation provided several important insights including the following:

- Watershed Phosphorus Loading Budget – The present average annual phosphorus load was determined to be excessive, based on comparison of a predicted load of 105 kg/yr to an acceptable load based on widely accepted estimation guidelines of 11 to 22 kg/yr. Thus, phosphorus loads to Lake Como must be reduced by approximately a factor of 5 (from 105 kg/yr to, at most, 22 kg/yr) before water quality improvements may be achieved. A large portion of the total phosphorus load, 53%, was estimated to come from storm drains.
- Evaluation of Restoration Alternatives – Restoration alternatives, featuring reduction or removal of storm drain loads, were evaluated using the watershed model. Removal or infiltration of storm drain flows from the system was predicted to result in 40% to 50% reductions in average annual loads of phosphorus to the ponds. Even if the additional phosphorus added to the soil through stormwater infiltration were to accumulate, the amount of groundwater added to the lake is relatively small. Additionally, minerals in the soil, and plants in the immediate area, will tend to keep the phosphorus from migrating far from the area of infiltration. Modifications that would take advantage of stormwater infiltration would have to be implemented along with additional modifications in order to achieve necessary water quality improvements.

Lake Como Restoration Recommendations

The water quality problems experienced by Lake Como can be resolved, but are not minor or easily repaired. A combination of projects conducted in-lake and in the watershed will be required to remove water quality impairment from the Lake Como system. The following 3 restoration tasks are recommended:

1. Repair leaky dam to support increased pond volume and water level.

2. Dredge nutrient-rich pond sediments to reduce sources of excess biological growth and increase pond volume.

3. Reduce nutrient loading to ponds through infiltration, detention, or removal of stormwater sources.

The Lake Como Restoration Study has successfully quantified present water quality and biological conditions in the system. The Lake restoration projects outlined above will, if implemented, result in dramatic improvements to the Lake Como system and are respectfully submitted for consideration.

1.0 INTRODUCTION

1.1 Overview

Lake Como (PALIS #52010) is located in the City of Attleboro and the Town of North Attleboro, Massachusetts within the Ten-Mile River Watershed (Figure 1-1). Lake Como is a small urban waterbody comprised of two small ponds connected by a culvert. The upstream pond, known as the West Pond, is smaller, approximately 2 acres in size, and is located in the Town North Attleboro. The downstream pond, known as the Main Pond, is larger, approximately 5 acres in size, and is located in the City of Attleboro. A roadway, Como Drive, passes between the ponds and an 18-inch culvert beneath the road connects West Pond and Main Pond. The downstream outlet of Main Pond is on the easternmost end of the pond and consists of an overflow weir to a culvert beneath Route 1. Outlet water from Lake Como flows into the Seven-Mile River in the City of Attleboro and eventually into the Ten-Mile River. Currently, many of the homes in the section of the Lake Como watershed occupied by the Town of North Attleboro are in the process of being sewered. Therefore, the septic systems that these homes originally depended on will be taken offline and wastewater will be transported out of the watershed for treatment. This process will hopefully improve the quality of the water in Lake Como by reducing nitrogen concentrations of groundwater that infiltrates into the lake. Appendix A contains photographs of the two ponds, culverts, and storm drain structures.

Lake Como is a eutrophic water body and is shallow and extensively vegetated during the summertime. Eutrophication is a process of nutrient accumulation and ecosystem change that occurs in aquatic ecosystems. This process occurs naturally as part of a long-term transition (e.g., from lake to marsh). Eutrophication can also occur culturally whereby the process is dramatically accelerated by the activities of man (McNaughton and Wolf, 1973). Problems associated with Lake Como eutrophication include dense populations of algae and rooted aquatic vegetation during the summertime. Additionally, low water levels have been observed during the summertime and have led to large areas of exposed lake sediment with associated foul odors due to decaying vegetation.

This Lake Como Restoration Study was undertaken by Massachusetts Department of Environmental Management, the City of Attleboro, and the Town of North Attleboro with assistance from The US Army Corps of Engineers. The purpose of this study is to characterize existing lake conditions and identify alternatives to restore the lake. This study focuses on characterization of present hydrologic, water quality, biological problems in Lake Como and an evaluation of restoration alternatives to restore the Lake Como system. The City of Attleboro and the Town of North Attleboro are concerned about hydrologic, water quality, and biological problems observed in Lake Como during the summertime; specifically, the excessive growth of aquatic vegetation that occurs each summer results in poor visual aesthetics and unpleasant odors. The ponds also experience low water levels for extended periods resulting a waterbody that, at times, is more akin to a wetland than a pond.

A goal of the Lake Como Restoration Study is to collect and apply data to support development of technically sound recommendations that will improve the water quality, habitat, and recreational utility of the lake. This report describes the data collection activities required to conduct a watershed-based pond restoration study, and presents the data collected and analysis of the results. In addition, an initial assessment checklist is provided in Appendix B and may be applied as a tool in conducting watershed-based pond assessment and restoration studies.

The Lake Como Restoration Study consists of two primary components, documented in this report. Firstly, a field investigation, designed and conducted to support characterization of present conditions in Lake Como, is described. A description of the field investigation, along with the results, is presented in Sections 2 and 3. Secondly, an evaluation of management alternatives was conducted, including a screening level watershed modeling application, and the development of restoration recommendations, which are presented in Section 4, 5, and 6.

1.2 Site Setting

The main tributary to Lake Como enters West Pond as a small, unnamed stream that originates in North Attleboro near Cushman Drive (Figure 1-2). This tributary stream forms the headwaters of the Lake Como watershed. The unnamed stream is approximately 3,000 feet in length and flows from west to east. The tributary passes through two detention systems, a 1-acre unnamed pond (upstream of the West Pond) and the West pond, prior to discharging into Main Pond.

Sources of water to Lake Como include the small-unnamed tributary stream, baseflow entering Lake Como from groundwater, and stormwater entering the lake during precipitation events. Stormwater enters the Main Pond primarily through three storm drains and one small, unnamed channel (Figure 1-2). These are:

- The Fuller Hospital storm drain at the south side of the main pond;
- The Esker Village subdivision storm drain immediately downstream of Como Drive (south of culvert);
- The Heather Street storm drain, named the North Attleboro drain for the purposes of this report, also located immediately downstream of Como Drive (north of culvert); and
- The small-unnamed channel at the north side of the main pond, named the Attleboro Drain for the purposes of this report.

Runoff from Route 1 enters the outlet flow immediately downstream of the overflow weir and therefore does not enter the main pond of Lake Como directly. Photographs of these main features are included in Appendix A of this report. Section 4 contains a discussion of historical changes to the watershed.

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Figure 1-1

Aerial Photograph of Lake Como and Surrounding Area

Attleboro and North Attleboro, Massachusetts

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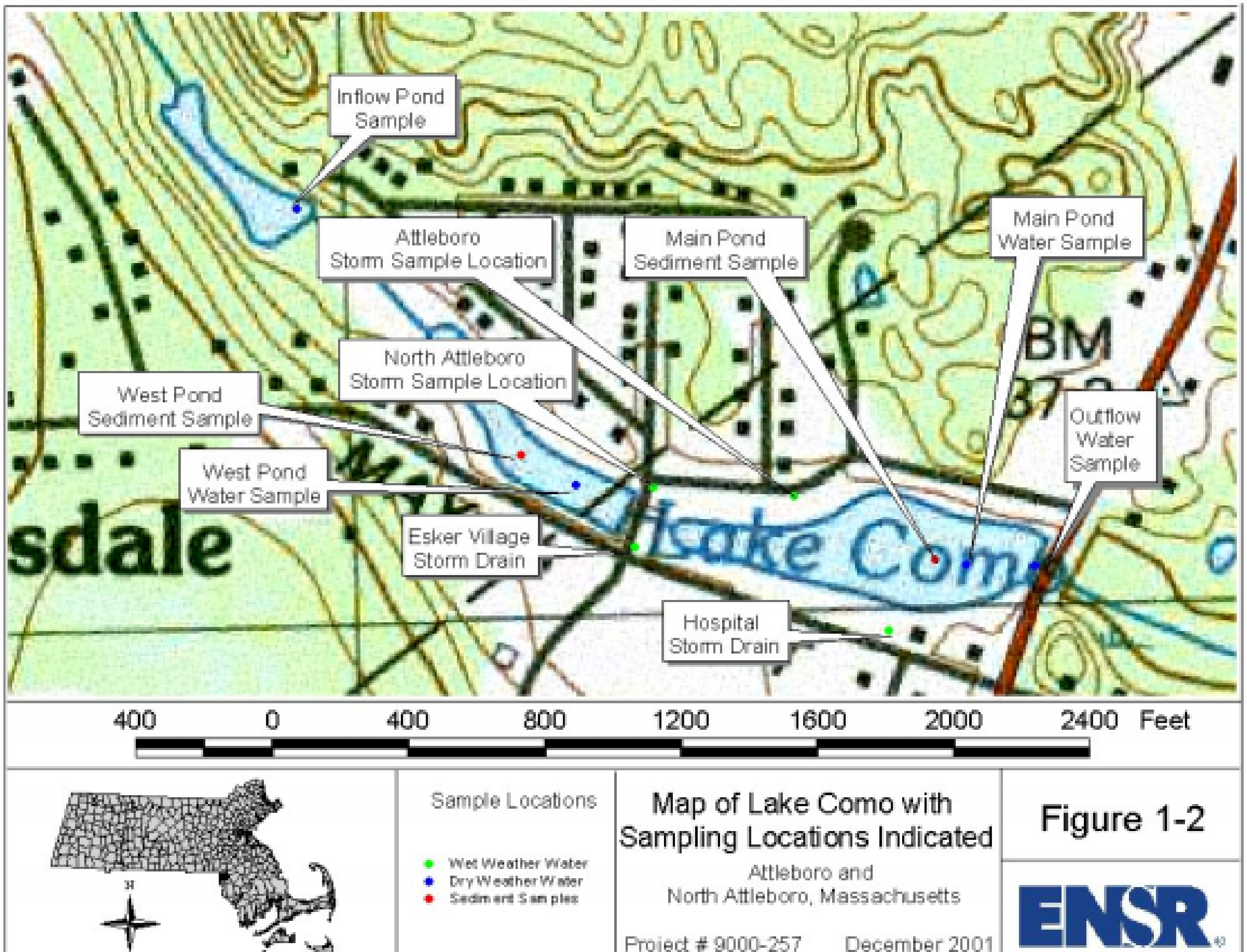

Map of Lake Como with Sampling Locations Indicated

Attleboro and North Attleboro, Massachusetts

Project # 9000-257 December 2001

Figure 1-2

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2.0 FIELD PROGRAM DESIGN AND METHODS

The Lake Como Restoration field program was designed to collect measurements necessary to support characterization of present conditions and evaluation of restoration alternatives to improve water quality in Lake Como. The field program design and methods employed to obtain measurements are described in this section. In addition, an initial assessment checklist is provided in Appendix B and may be applied as a tool in conducting watershed-based pond assessment and restoration studies.

2.1 Field Program Design Approach

The field program was designed to collect sufficient hydrologic, water quality, sediment, and biological data to support the Lake Como Restoration Study goals. The field program design included water quality/hydrology surveys designed to capture nutrient loads within the ponds and in waters entering the ponds under various conditions. Nutrient loading estimates were obtained by analyzing nutrient concentration and streamflow measurements. A sediment survey was conducted to characterize sediment volume and quality. Biological surveys were conducted to characterize the nature and extent of aquatic biology in the ponds during the summertime. Measurements were collected during a total of 8 field survey events in support of the Lake Como Restoration study. The 8 field survey events featured collection of hydrologic, water quality, sediment, and biological measurements and may be represented as follows:

- *Water Quality/Hydrology Surveys:*
  - *Dry-weather surveys (4) – conducted Sept. 2000, May 2001, and August 2001 (2)*
  - *Wet-weather surveys (3) – conducted September 2000, June 2001, and June 2002*
  - *Pond bathymetry/sediment survey (1) – conducted September 2000*
- *Biological Surveys*
  - *(1) Aquatic Macrophyte Survey – conducted September 2000*
  - *(4) Phytoplankton Survey – conducted September 2000, May 2001, and August 2001*
  - *(2) Zooplankton Survey – conducted May and August 2001*

A description of each survey type, including survey objectives, sampling activities, and sampling methods is provided below.

2.2 Water Quality/Hydrology Surveys

Dry-weather and wet-weather surveys are hydrologic were water quality surveys conducted under different conditions. The field program design and sampling methods for dry-weather and wet-weather water quality/hydrology surveys are described below.

2.2.1 Dry-Weather Survey Design

Dry-weather surveys were performed to assess water quality in Lake Como and that of water entering the ponds from the watershed. Water quality measurements were collected at the inlet, in the West and Main Ponds, and from storm drains, if flowing. Water quality measurements were analyzed to support characterization of the overall nutrient budget. Nutrient loads from storm drains, representing watershed subbasin areas, were measured to support quantification of non-point source loads throughout the year.

Hydrologic data collection focused on flows and average water velocity estimates made at the inlet, the culvert between the West and Main Ponds, at the outlet, and where flow was observed at the storm drains.

Water quality data collection included in-situ water quality measurements of temperature, dissolved oxygen concentration, pH, conductivity, and grab sampling for laboratory analysis of nutrient-related chemical parameters. These measurements were made at the inlet, the West Pond, the Main Pond, and at the outlet.

2.2.2 Wet-Weather Survey Design

Wet-weather surveys were performed to measure nutrient loads from storm drains tributaries during storm events. Non-point source nutrient loads are highly variable over time. In general, non-point source nutrient loads increase dramatically during precipitation events as overland and subsurface flows carry nutrients to a receiving waterbody. Wet-weather non-point source nutrient loads were measured at the storm drains to evaluate the peak nutrient loads to the Lake Como system. Specifically, wet-weather surveys were designed to capture nutrient concentrations in storm drains discharging to Lake Como during the rising limb of storm hydrographs that were induced by precipitation events. By capturing storm induced nutrient concentrations in tributaries, nutrient loads from overland flow may be estimated and determinations made regarding the relationship between nutrient loads and land use practices within the tributary watersheds.

Three wet-weather water quality surveys were completed during precipitation events in mid-September 2000, mid-June 2001, and late June 2002. Wet-weather survey methods featured deployment of automated water sampling equipment at each of the four drains (North Attleboro, Attleboro, Hospital, and Esker Village). At each location grab samples were collected for laboratory analysis of nutrient-related chemical parameters.

2.2.3 Hydrologic Data Collection Methods

Field sampling methods employed in the collection hydrologic measurements during the dry-weather and wet-weather water quality/hydrology surveys are described in this section. In general, field sampling crews and sampling equipment were mobilized from ENSR's Westford, MA office for all surveys. Equipment used included a vehicle, a canoe, a water quality meter, calibration solutions, coolers containing water sample bottles and ice, a first-aid kit, and a cellular phone.

Hydrologic measurement of streamflow (i.e., volumetric flow rate) was typically estimated based on two methods.

- Time of Travel Estimation. A 3-foot long reach was identified with approximately uniform flow. Small leaves or twigs were allowed to float through this reach. The time for the object to pass through the reach was measured and noted. In this way, a water velocity was established. This velocity, multiplied by the width and the depth of the water in the reach, yielded a volumetric flow rate.
- Direct volumetric flow measurement. A container of known volume was set in the path of the flowing water to capture flow during a measured period of time. The time for the container to fill (or partially fill) was recorded, along with the volume filled. The volume divided by time yielded a volumetric flow rate. This was particularly useful for measuring flow at the outfall pipe.

Traditional methods, using either a pygmy rotating cup current meter or a Marsh McBirney electro-magnetic meter, in accordance with guidance provided by the United States Geological Survey, was also used.

2.2.4 Water Quality Data Collection Methods

Field sampling methods employed in collecting water quality measurements during the dry-weather and wet-weather water quality/hydrology surveys, are described in this section. In general, field sampling crews and sampling equipment were mobilized from ENSR's Westford, MA office for all surveys. Equipment used included a vehicle, a canoe, a water quality meter, calibration solutions, coolers containing water sample bottles and ice, a first-aid kit, and a cellular phone.

Two primary water quality data collection methods were employed in the Lake Como watershed including in-situ water quality measurements and laboratory analysis of water samples for nutrient-related parameters. Each water quality method is described below.

2.2.4.1 In-situ Water Quality Measurements

In-situ measurements of temperature, dissolved oxygen concentration (and % saturation), pH, and conductivity were collected using portable field equipment. Temperature and dissolved oxygen were measured with a YSI 6820; the dissolved oxygen was calibrated prior to use.

2.2.4.2 Water Sample Collection for Laboratory Analysis (Dry-Weather)

Water samples were collected for laboratory analysis at four sampling locations throughout the study area. Water samples were placed in sample bottles prepared and provided by the laboratory. All samples were labeled with information including the project name, sampling time and date, and the sample location. Samples were collected and labeled in a manner that uniquely identified each individual sample bottle. Once filled, sample bottles were put in a cooler filled with ice. Samples were kept cold and were sent by FedEX to the analytical laboratory within 4 hours of sample collection to comply with the shortest sample holding time of 6 hours for fecal coliform.

Water samples were collected by boat in the West and the Main Ponds. At the inlet, field personnel waded to several feet offshore before selecting a sampling location. At the outlet, water was taken from the outfall. Except for the fecal coliform bottles, which contained a preservative, the bottles were rinsed with water prior to collecting the sample. Once collected, the bottles were labeled with all pertinent information.

During dry-weather sampling, water samples were analyzed for the following parameters: alkalinity, nitrate-N, ammonia-N, Total Kjeldahl Nitrogen (TKN), total phosphorus, dissolved phosphorus, turbidity, total coliform, fecal coliform, fecal streptococcus, total suspended solids (TSS), and total dissolved solids (TDS), at a state certified laboratory.

2.2.4.3 Water Sample Collection for Laboratory Analysis (Wet-Weather)

Wet-weather grab samples were collected using simple automated grab samplers. Wet-weather grab samples were analyzed for turbidity, chloride, alkalinity, total phosphorus, TKN, nitrate-N, and ammonia-N (bacteria sampling was not conducted during wet weather due to the use of automated samplers and sample holding times).

The automated grab sampler design is shown in Figure 2-1 and consists of a sample bottle equipped with a stopper with two tubes, one shorter (to allow water to enter) and one longer (to allow air to escape). The sample bottle was attached to a wooden stake that was pounded into the channel bed within the path of the flow. The bottle was attached to the stake such that the shorter tube was approximately one inch above the water line (depending on the characteristics of the tributary cross-section). When the water level in the river rose due to storm water runoff, the sample bottles were filled with water.

The samplers were retrieved shortly after being filled. Sample bottles were put in a cooler filled with ice. Samples were kept cold and delivered to the analytical laboratory.

2.3 Sediment Survey Design and Methods

The design of Lake Como sediment surveys and sampling methods employed are described below.

2.3.1 Sediment Survey Design

A pond bathymetry and sediment quality survey was performed to support characterization of sediments and potential sediment impacts on water quality at Lake Como. A bathymetric and sediment thickness survey was performed in the West and Main Ponds. Bathymetry measurements were used to support estimation of impoundment volume and average residence time. Sediment thickness measurements were used to support assessment of sediment impacts on lake water quality.

Sediment quality sampling was conducted in both West Pond and Main Pond. Sediment sampling was conducted to evaluate the impact of impoundment sediments on the nutrient budget of the lake system. Sediment sampling was also completed to provide a preliminary toxicologic characterization of the sediments to support a dredging feasibility evaluation. The feasibility of dredging and the disposal alternatives for dredged sediments are highly dependent on the sediments' characteristics. Methods employed to collect sediment samples are described below.

2.3.2 Pond Sediment Thickness and Bathymetry Survey Methods

The water and sediment thickness surveys were boat-based and involved collection of measurements across transects to support bathymetric and sediment thickness mapping. Locations of measurements were identified using landmarks and recorded on topographic maps (Figure 2-2). A Geographic Positioning System was not used for the survey.

Water depth was estimated by probing the water and sediment column using a graduated pole. The same pole was then forced through the sediment until first refusal (rock, tight sand, gravel or clay) and the water depth was subtracted from the total depth to obtain sediment thickness.

2.3.3 Sediment Quality Sampling Methods

Sediment sampling was conducted in both the West and Main Ponds, concurrent with the sediment thickness evaluation. Samples were collected using an Eijkelkamp sediment core sampler to a sediment depth of approximately 2 feet. Samples were brought to the surface and aggregated in a clean plastic bucket until enough volume was collected to meet laboratory requirements. Once collected, sediment was immediately placed in laboratory provided jars and stored on ice. A courier picked up the sediment samples for delivery to the state certified laboratory within a few hours of collection.

2.4 Biological Survey Design and Methods

The design of Lake Como biological surveys and sampling methods employed are described below.

2.4.1 Biological Survey Design

An assessment of the aquatic plant community (macrophytes) in Lake Como was conducted to determine species diversity and density, and to document the presence of nuisance vegetation. The evaluation of aquatic macrophytes, phytoplankton, and zooplankton provides insights into the trophic condition of a waterbody. It is also important to consider macrophyte and algal growth, since an overabundance of macrophytes and/or algae can threaten water quality due to diurnal oxygen cycles that can swing from an over saturation of oxygen during the afternoon to a depletion of oxygen prior to sunrise. An overabundance of macrophytes and algae are also important because they can change the species composition of fish in the waterbody and diminish recreational uses.

Biological data collection was conducted as part of the dry-weather investigations and focused on determination of the types of aquatic vegetation present in the system and their distribution in the West and Main Ponds during summertime conditions.

2.4.2 Biological Sampling Methods for Phytoplankton Assessment

Plankton samples were collected at the same stations as the West and Main Pond surface water samples. As with the water samples, the phytoplankton samples are meant to be representative of the entire water body. These samples were preserved with lugols solution, concentrated by settling, as needed, and viewed in a Palmer-Maloney counting chamber at 400X magnification with phase contrast optics. Between the concentration and the area scanned for identification/counting, the multiplication factor (cells recorded to cells/ml) is <50, usually <20. Counting proceeded until each successive strip does not change the ratio of the dominant algal types (those comprising >50% of all cells cumulatively) by more than 10%.

2.4.3 Biological Sampling Methods for Aquatic Macrophyte Assessment

Macrophyte assessment is primarily based on visual examination of the overall pond habitat. Its purpose is to determine the range of aquatic plant types in the system and relative dominance by coverage or frequency of occurrence. Macrophyte assessment was performed as follows.

1. Aquatic plant distribution and density was surveyed on September 7, 2000. Maps were created to illustrate distribution by species, overall percent cover, and the portion of the water column occupied by aquatic macrophytes.

2. Plants were identified to the species level in the field or lab according to Hellquist and Crow (1980-1985).

3. Plant cover was estimated on a scale of 0-4 as follows:

   - 0: No cover, plants absent
   - 1: 1-25% cover
   - 2: 26-50% cover
   - 3: 51-75% cover
   - 4: 76-100% cover

4. Plant biomass was estimated on a scale of 0-4 as follows:

   - 0: No biomass, plants absent
   - 1: Low biomass, plants growing only as a low layer on the bottom sediment
   - 2: Moderate biomass, plants protruding into the water column, but rarely reaching the surface and not at nuisance densities
   - 3: High biomass, plants filling more than half the water column and often reaching the surface, nuisance conditions and/or habitat impairment perceived
   - 4: Extremely high biomass, water column filled and/or surface completely covered, nuisance conditions and habitat impairment severe.

2.5 Field Program Methods: Quality Assurance Program

All sampling was carried out in order to assure sample precision, accuracy, and representativeness. Precision is a measure of the degree to which two or more measurements are in agreement, and was assessed through the determination of duplicate samples, collected or measured randomly, representing about 29% of the actual number of samples. Precision was measured as the relative percent difference (*RPD*) between sets of values:

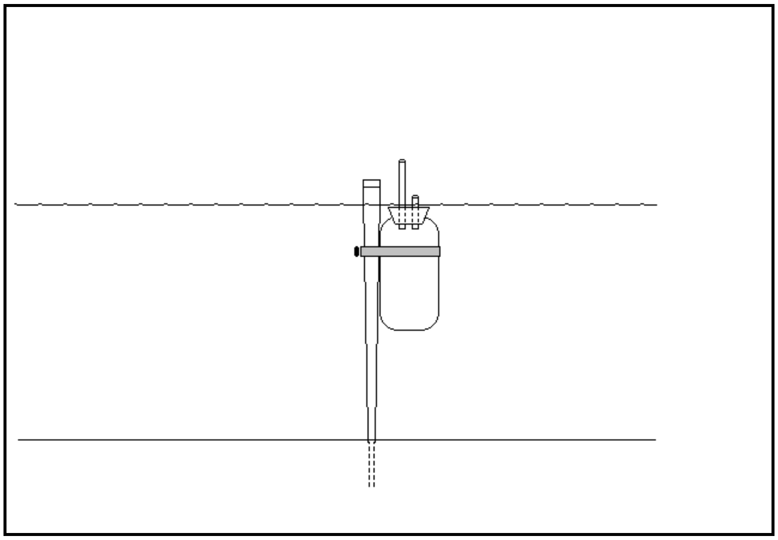
$$RPD = \frac{(Amount\ in\ Sample\,1 - Amount\ in\ Sample\,2)}{0.5\,(Amount\ in\ Sample\,1 + Amount\ in\ Sample\,2)}\,x\,100$$

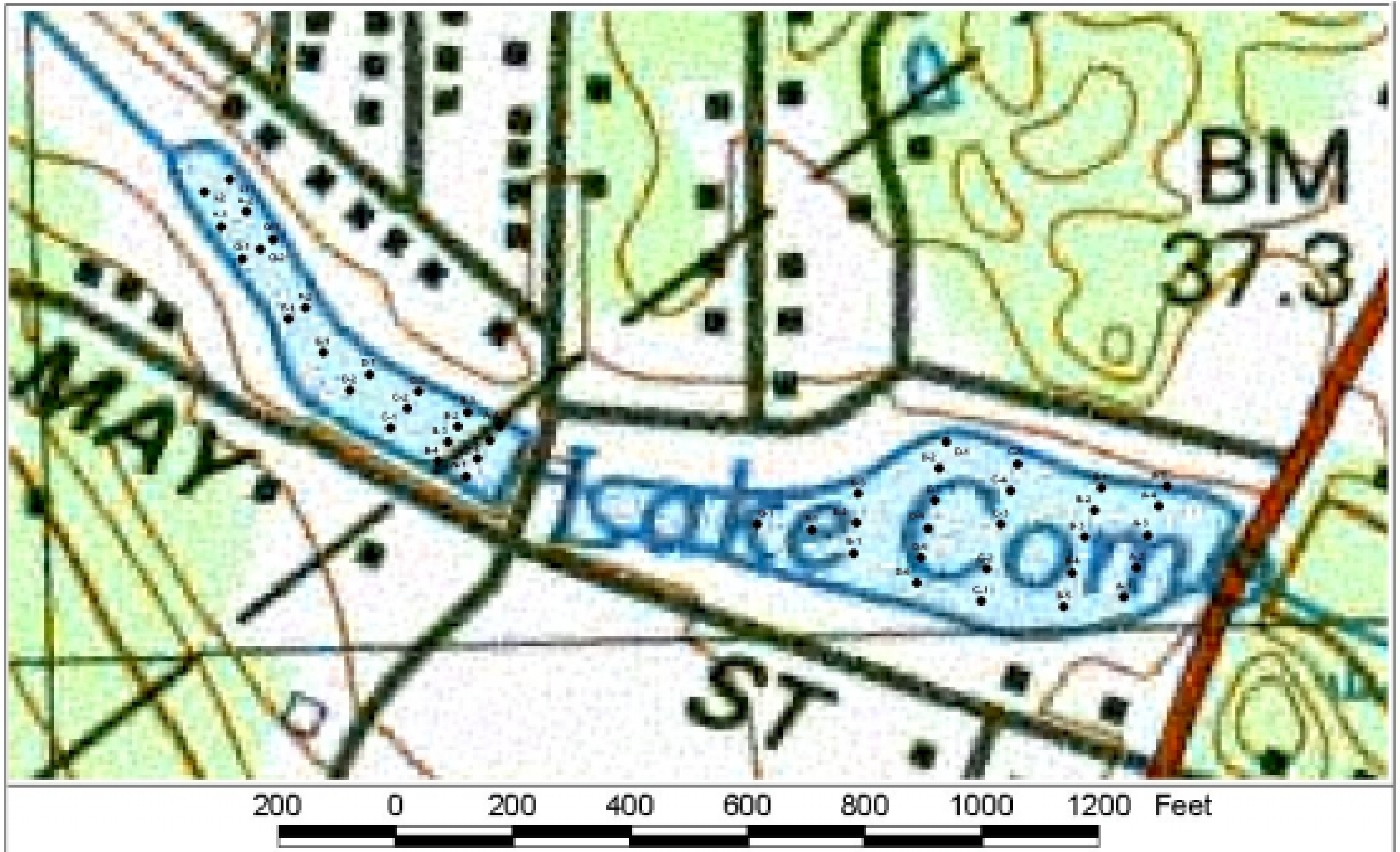
Two outlet duplicates, two West Pond duplicates and two inlet duplicates were collected during the sampling period September 2000 through June 2002. RPD values for water quality ranged from 0.7% to 27%, depending on the parameter, with RPD values higher than about 15% resulting from small differences in results near the detection limit for several parameters.

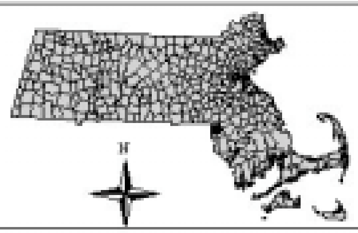
Accuracy is the degree of agreement between the observed value (i.e., measured, estimated, or calculated) and an accepted reference or true value (i.e., the real value). Accuracy was achieved through the adherence to all sample collection, handling, preservation, and holding time requirements, but was not tested with blanks or spikes in this study. The laboratories employed to analyze samples are certified by the Commonwealth.

Representativeness expresses the degree to which data accurately and precisely represent a characteristic of a parameter, process, population, or environmental condition within a defined spatial and/or temporal boundary. Representativeness of the data collected was maximized by following the study design and applying the proper sampling techniques and analytical testing. Where choices of stations to be sampled were made, effort was expended to ensure that those sites sampled were representative of the conditions the study intended to assess.

Figure 2-1 Schematic Diagram of Automated Wet-Weather Grab Sampler







Map of Lake Como with Transect Sampling Locations Indicated

Attleboro and
North Attleboro, Massachusetts

Project # 9000-257   December 2001

Figure 2-2

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3.0 FIELD PROGRAM RESULTS

Lake Como Restoration Study field program results are provided below. A description of the study area and key features is also presented, based on a field reconnaissance survey. The results of dry-weather and wet-weather survey are then presented including all hydrologic and water quality measurements. Lastly, the results of sediment and biological surveys are presented.

3.1 Description of Study Area and Key Features

The Lake Como study area is shown in Figure 3-1 with photographs of key features. The Lake Como watershed is approximately 200 acres is size and is located both Attleboro and North Attleboro, Massachusetts. Lake Como is comprised of two small ponds, West Pond and Main Pond, connected by an 18 inch culvert along Como Drive - the town border (see Figure 3-1, photo #6). The upstream West Pond, is approximately 2 acres in size and is located in North Attleboro (Figure 3-1, photo #7). The downstream, Main Pond is approximately 5 acres in size and is located in Attleboro. The downstream outlet of main pond is a dam with an overflow weir to a culvert beneath Route 1 (Figure 3-1, photo #3). Outlet water from Lake Como flows into the Seven-Mile River in Attleboro and eventually into the Ten-Mile River.

Identification and characterization of the quantity and water quality associated with water entering the two ponds at Lake Como are important components of the study. The following sources of water to Lake Como have been identified:

- Small-unnamed tributary stream flowing into West Pond from the west;
- Groundwater baseflow;
- The Fuller Hospital storm drain along the southern side of Main Pond (see Figure 3-1, photo #4);
- The Esker Village storm drain along southern side of Main Pond(see Figure 3-1, photo #5);
- North Attleboro storm drain located along northern side of Main Pond (see Figure 3-1, photo #1);
- Attleboro storm drain located along northern side of Main Pond (see Figure 3-1, photo #2)

Structural characteristics of the culverts, storm drains, and dams can affect movement of water and, indirectly, affect pond water quality. Several important observations were made of the characteristics and condition of physical structures in the Lake Como study area. Each observation of key physical structural conditions is presented below.

3.1.1 Leak in Outlet Dam Structure

A continuous leak was observed in the outlet dam at Route 1 (see Figure 3-1, photo #3). The dam is a concrete structure and is cracked at a point near its base. As a result, water continuously flows out the Main Pond at the dam. The water level was consistently observed to be at a level below the weir overflow level during the study. The leaky dam effectively lowers the water level and reduces the storage capacity of Lake Como. Lower water levels can result in increased areas of exposed sediments and increased growth of rooted aquatic vegetation. Smaller water volumes result in diminished residence time for water in the system, potentially decreasing the ability of the ponds to support fish populations.

3.1.2 Storm Drain Overland Flow

The Attleboro, North Attleboro, and Fuller Hospital storm drains (see Figure 3-1, photos #1, #2, and #4) do not flow directly into the surface water of Lake Como. During the period of the study, waters from each of these storm drains had to flow approximately 100 feet across dry land prior to reaching the pond. Based on field observations, water draining from Fuller Hospital to the Main Pond traveled along a gradual downhill slope. The storm drains on the northern side of Main Pond (Attleboro and North Attleboro drains), however, appeared to have to flow over minor berms to reach the Main Pond.

Overland flow, along the path of storm drainage, is expected to result in the infiltration of storm water to into the subsurface during storm events, thus reducing the total water volume reaching the pond. In the case of the Attleboro and North Attleboro drains, stormwater may reach the Main Pond only during major precipitation events when sufficient stormwater flow is present to enable overland flow.

3.2 Dry-Weather Water Quality Survey Results

Dry-weather sampling surveys were conducted on September 7, 2000, May 29, 2001, August 1,2001, and August 30, 2001. Table 3-1 contains a compilation of dry-weather water quality data. A summary of dry-weather data is provided below for nutrients, bacteria, and other parameters.

3.2.1 Nutrient Data

<u>Overview</u>

Nitrogen and phosphorus are essential nutrients for plant growth. High concentrations of nitrogen and phosphorus in the water column provide an ideal environment for aquatic biological growth. Although phosphorus tends to be the limiting nutrient in freshwater systems, high nitrogen concentrations indicate a fertile aquatic environment. There are several forms of nitrogen but only some are available for uptake by aquatic organisms. Ammonia and nitrate are the two forms of nitrogen most accessible to aquatic vegetation; organic nitrogen is bound up in organic material and is unavailable. Organic nitrogen is indirectly measured by taking the difference between TKN and ammonia.

Currently, there are no numerical surface water standards for nutrients in Massachusetts; however, such standards are presently being developed. Acceptable ranges for nitrate nitrogen in this ecoregion are estimated to be between 0.3 – 0.6 mg/l, with < 0.3 mg/l ideal (Wetzel, 1975). Nitrate values between 0.6 – 1.0 mg/l are indicative of deteriorating aquatic environment and > 1.0 mg/l indicates a poor aquatic environment or highly eutrophic conditions (Wetzel, 1975). Levels of ammonia nitrogen greater than 1.0 mg/L are generally considered high while concentrations less than 0.1 mg/L are considered low. Similarly, levels of TKN greater than 3.0 mg/L are generally considered high while levels less than 0.3 mg/L are considered low.

For phosphorus, acceptable concentrations are estimated to be less than 0.03 mg/l. Values above 0.03 mg/l are associated with environments where biotic productivity can reach nuisance levels (Wetzel, 1975). Phosphorus concentrations above 0.05 mg/l are estimated to be sufficient to support eutrophication and concentrations above 0.10 mg/l are extreme and water quality impairment in lakes under those conditions is believed to be inevitable.

Results

Table 3-1 contains a summary of nitrogen and phosphorus concentration measurement collected during the dry-weather surveys. Total nitrogen (TKN) concentrations ranged from 0.3 to 4.0 mg/l. Nitrate concentration measurements were low to moderate (<0.01 – 0.12 mg/L), with the exception of a relatively high measurement collected at the Esker Village storm drain on May 29, 2001 (1.62 mg/l). Ammonia values were low to moderate at all stations (<0.01 – 0.12 mg/L).

Total and dissolved phosphorus concentration measurements collected in the Lake Como system were indicative of eutrophic systems. Total phosphorus values were moderate to high ranging from 0.03 to 0.31 mg/L, with the higher measurements collected at the inlet and in the Main Pond. Dissolved phosphorus concentration measurements ranged from 0.02 to 0.05 mg/L. Nitrogen and phosphorus concentration measurements were observed to be excessive and more than sufficient to support eutrophic conditions.

3.2.2 Bacterial Data

Overview

Fecal coliform (FC) and fecal streptococci (FS) are bacterial indicators for potentially harmful pathogens. Fecal coliform analyses measure bacteria present from wastes of human and other warm-blooded animal sources, such as ducks, raccoons and family pets. FC can multiply in the environment and may be sustained over time in a waterbody. FC is the regulatory parameter for potentially harmful pathogenic bacteria in the State of Massachusetts; however, not all bacteria included in fecal coliform counts are harmful.

Bacterial colony measurements are typically highly variable. As a result, intensive sampling and a statistical analysis of bacterial data are required to support rigorous bacterial characterization. Massachusetts State Water Quality Standards state that fecal coliform bacteria in Class B waters shall not exceed a geometric mean concentration of 200 organisms per 100 milliliters (ml) in any representative set of samples. Furthermore, not more than 10% of the samples shall exceed a concentration of 400 organisms per 100 ml.

Fecal streptococcus is also found in the digestive systems of humans and other warm-blooded animals and is another indicator of possible harmful pathogenic contamination. FS sampling and analyses were conducted because FS, unlike FC, do not multiply in waterbodies and are not sustained for extended periods. Thus, measurement of FS may provide a more accurate characterization of bacteria recently introduced to the waterbody. Also, the ratio of FC to FS has been successfully applied to support the identification of the source of bacteria – enabling differentiation between sources (e.g., bird vs. human waste).

Results

Fecal coliform measurements ranged from 18 to 2,400 colonies per 100 ml. The high levels of FC measured in August 2001 were probably the result of a very long residence time in the lake and the accumulation of waterfowl that were grazing on the aquatic macrophytes in and around the perimeter of the lake. Fecal streptococcus numbers ranged from 10 to 40 colonies per 100 ml. Levels of FS bacteria were too low to apply the FC:FS ratio to evaluate potential sources of bacteria. In general, bacteria levels were observed at levels typical of impoundments during the summer season.

3.2.3 Other Water Quality Data

Dissolved oxygen enters the aquatic environment through diffusion from the air and from plants through photosynthesis. Dissolved oxygen is essential for metabolic activities that occur in aerobic biota. In addition to biological importance, dissolved oxygen concentrations strongly affect the solubility of inorganic nutrients. The Massachusetts water quality standard for dissolved oxygen concentration is 5.0 mg/L. Dissolved oxygen concentrations measured in Lake Como ranged from 1.0 to 9.7 mg/L. Violations of the water quality standard (i.e., values below 5.0 mg/L) were observed in both ponds, at the inlet and at the outlet. Temperature in the Main and West Ponds ranged from 17.7 to 27.4°C. The warmest value was recorded in the Main Pond on August 1, 2001. The inlet and outlet had similar values (16.3 – 22.6 °C), although slightly cooler.

The pH is a measure of the hydrogen ion concentration in water. The pH scale ranges from 0 to 14; zero being a highly acidic solution and 14 being highly alkaline, 7 being neutral. The range of pH in a majority of open waters ranges from 6 to 9 (Wetzel, 1983). Although most organisms have developed the ability to adapt to minor fluctuations in pH, or have evolved in extreme acidic or alkaline environments, sudden shifts in pH can be detrimental to organisms in the aquatic environment. Such conditions can occur during stormwater runoff events and/or wastewater discharges. The Massachusetts Water Quality Standards state that pH should fall between 6.5 and 8.3 standard units (SU) and there should not be a change of 0.5 SU from background conditions for Class B waters. Values for pH ranged from 6.4 to 8.4 SU for all sampled stations in the Lake Como study. The West and Main Pond values ranged from 6.6 – 7.2 SU and were within the acceptable range.

Alkalinity is a direct measure of the concentration of compounds such as bicarbonates, carbonates, and hydroxides in a solution. This concentration of “neutralizing” material determines the aquatic system’s ability to buffer against acidic inputs. There is no state or federal standard for alkalinity. However, alkalinity values above 20 mg/l are usually indicative of a system with an adequate buffering capacity against acidic inputs, such as acid rain, wastewater and stormwater discharges (Godfrey, 1988). Values ranging from 2 to 80 mg/L are typical in Massachusetts (PALIS Database, MA DEP). Alkalinity ranged from 41 – 77 mg/L for all sampled stations with the highest value recorded at the Esker Village storm drain.

Chlorides are naturally occurring salts such as sodium chloride (NaCl) and magnesium chloride ($MgCl_2$). Chlorides are also required for plant and animal cell function. Elevated chloride levels occur due human activities such as the application of road salt and waste disposal. There is no state standard for chloride. However, chloride levels above 10 mg/L are undesirable and levels above 100 mg/L will likely impact water quality (McKee and Wolf 1963). Chloride levels within the Lake Como system ranged from 19.0 to 57.3 mg/L. Conductivity is a measure of the soluble mineral or salt content of water and is used as an indicator of an aquatic system’s potential for fertility. Values exceeding 200 mg/L indicate a fertile environment. Specific conductivity were slightly elevated (155 – 346 mg/L, mean 234 mg/L).

Turbidity is a measure of water clarity. Turbid waters are indicative of high levels of suspended particles that may include algal cells, silt, or resuspended sediments, and are usually associated with poor water quality. Acceptable standards depend on water body use, but turbidity readings higher than 10 nephalometric turbidity units (NTU) are indicative of potentially undesirable water quality. Most “clean” New England lakes exhibit turbidity ranging from 1 to ~5 NTU. Turbidity ranged from 1.5 to 24.0 NTU in the Lake Como study, with the highest value recorded at the inlet on August 1, 2001. Turbidity values in the Main and West Ponds were generally below 5.0 NTU with the exception of the Main Pond on September 7, 2000.

Secchi disk measurements were made at several locations within both the west and main ponds. The Lake bottom was visible during the survey because the water level in the Lake was shallow. Thus, the Secchi depth was greater than the water depth during the survey.

3.3 Wet-weather Water Quality Survey Results

Wet-weather sampling surveys were conducted on September 13, 2000, June 17, 2001, and June 27, 2002.. Table 3-2 contains a compilation of wet-weather water quality data. A summary of wet-weather data is provided below for nutrients and other parameters.

The wet-weather event of September 13, 2000 generated more than 0.25" of rain, but wet-weather samples were only collected at two of the four locations (the North Attleboro and Esker Village drains) where storm samplers were set. The sample bottles may not have filled at those locations due to lack of overland flow necessary for stormwater to reach the Lake from those locations. Also, 0.25" is a relatively small precipitation event and may not have been sufficient, given the small size of the associated subbasins, to support stormwater flow.

The June 17, 2001 storm event produced 7.14 inches of rain (recorded at Pawtucket, RI) between approximately noon and 9 PM. Samples were collected at two of four locations, the Attleboro residential area drain and the Fuller Hospital area storm drain. It is suspected that due to the magnitude of the rain event, the samplers at the other locations were washed away before they could fill. They were retrieved several tens of feet downstream of their set locations.

The June 27, 2002 storm event resulted in 0.13 inches of rain (recorded at Providence, RI). Stormwater samples were collected at one of four locations (Esker Village drain). The storm event did not appear to have generated enough runoff to fill the other sample bottles.

3.3.1 Nutrient Data

Nitrogen levels were moderate to high throughout this investigation and suggest substantial loading from the watershed. Nitrate levels were high with all values exceeding 1.0 mg/L (1.2 – 2.44 mg/L). Ammonia values were moderate to high (0.10 – 1.01 mg/L). Total Kjeldahl nitrogen (TKN) values ranged from moderate to high (0.71 – 7.3 mg/L); the high values (>3.0 mg/L) were recorded at the Attleboro and Fuller Hospital drains. Total phosphorus values were high, with values ranged from 0.06 to 6.20 mg/L. The higher phosphorus values were recorded at the Attleboro and Fuller Hospital drains.

3.3.2 Other Water Quality Data

Alkalinity and chloride concentrations were lower during wet weather survey while turbidity, total dissolved solids and total suspended solids concentrations were higher. Alkalinity ranged from 8 – 56 mg/L for all sampled stations. The lowest value was recorded at the North Attleboro storm drain. All other values were above 20 mg/L. Lower alkalinity is often associated with surface water runoff due to the acidity of the precipitation. Chloride concentrations were lower during wet weather sampling, which suggests that there is a constant source that becomes diluted during wet weather. Turbidity, total suspended solids and total dissolved solids were elevated at all stations suggesting that storm water catch basins, and any other sediment collection devices, are not functioning properly in this system. All storm inputs were relatively similar indicating that stormwater quality problems are diffuse and not limited to one drainage area.

3.4 Sediment Survey Results

Results of pond bathymetry, sediment thickness, and sediment quality assessments are provided below.

3.4.1 Pond Bathymetric Data

The bathymetry indicated that both the West and Main Ponds are relatively shallow with typical water depths of 1 to 2 feet. Figure 3-2 presents water depth contours for both ponds. The water depths in the Main Pond were mostly less than 3 feet and only a small section of the pond had depths greater than 4 feet. In both ponds, the deepest sections were near the outlets. Additionally, many of the areas where the water was less than 1 foot deep, were largely exposed mud surfaces that were overgrown with rooted aquatic vegetation. At the time of the survey, the surface area of the West Pond was 1.9 acres and the surface area of the Main Pond was 4.9 acres, for a total of 6.8 acres. The water volume in the West Pond was estimated to be 2.2 acre-ft and the volume in the Main Pond was 8.4 acre-ft for a total estimated water volume of 10.6 acre-ft.

The surface of the Main Pond was actually 2 feet 8 inches below the top of the overflow weir at the outlet structure, at the time of the bathymetry survey. This indicates that the lake could be almost 3 feet higher when surface water is available to support the increased water levels. If filled to the top of the overflow weir, the volume of Lake Como would be at least 28.7 acre-ft. A discussion of the benefits of increasing the summertime pond volume by maintaining the integrity of the outlet structure is presented in Section 6.3.1.

The downstream lake outflow structure presently leaks, allowing water to leave the waterbody as underflow through the weir structure. Therefore, even at the low lake level measured during the investigation, water was flowing out of the pond from a level approximately 4 feet down from the top of the weir. This appears to be unintentional result of a crack near the base of the structure. This leak allows the lake to drain regardless of the lake level relative to the top of the weir.

3.4.2 Sediment Thickness Data

Soft sediments in the West and Main Ponds were found to be typically 1 to 2 feet thick. The thickness of soft sediment in the West Pond was thickest near the upstream end (Figure 3-3). This distribution suggests that much of the sediment entering the West Pond settles near the inlet where suspended material in stream flow is removed as water velocity and carrying capacity decrease. Throughout much of the West Pond the sediment thickness is from 1 to 2 feet with a thickness greater than 3 feet occurring over a relatively small area at the upstream end of the lake. Sediment thickness in the Main Pond is of a similar magnitude, if not slightly less. Sediment thickness in the Main Pond generally does not exceed 2 feet and is generally between 1 and 2 feet. The sediment thickness maps developed as part of this investigation were used to determine the volume of soft sediment in Lake Como. The soft sediment volume in the West Pond was calculated to be 4,830 cubic-yards and the soft sediment volume in the Main Pond was calculated to be 7,850 cubic yards, for a total of 12,680 cubic yards of soft sediment.

3.4.3 Sediment Quality Assessment

The sediment assessment in Lake Como was undertaken to evaluate the quality of soft sediment in relation to dredge disposal guidelines. Sediment nutrients were also analyzed to support evaluation of the role of pond sediments in the overall nutrient budget.

Dredging is one potential restoration option for Lake Como and the sediment quality assessment supports evaluation of dredging alternatives. Once removed from the lake, sediments typically become part of upland soils and are subject to the Massachusetts Contingency Plan (MCP) restrictions. High concentrations of pollutants in lake sediments do not necessarily prevent dredging activities, but they do limit dredged material disposal options, which in turn increases the costs associated with disposal.

Metals concentrations in sediment are typically compared to the 90th percentile in the Massachusetts Department of Environmental Protection (MA DEP) Background Soil Data Set and the MCP Reportable Concentration Soil-1 category (MCP RCS1, the most stringent soil category) to determine dredging feasibility. It is not uncommon that pond sediment concentrations exceed the MADEP 90th percentile for soil; the mean values for several metals in pond sediments in Massachusetts exceed the background soil conditions (Table 3-3). Pond sediments are the ultimate repository for many metals within a watershed, and appear to be quite inert once incorporated into those sediments. However, disposal of sediments, with concentrations higher than background soil conditions, requires disposal precautions and other limitations. Options for disposal of sediments with concentrations in excess of the MCP RCS1 standards are limited and associated costs may outweigh any benefit gained from removal.

Metal concentrations did not exceed the MCP RCS1 standard in either sample collected in Lake Como (Table 3-3). The West Pond sediment sample could have exceeded the MA DEP Background Soil Data Set 90th percentile for nickel since, in this case, the detection limit was too high. However, this is not likely since the downstream sample indicated less nickel than the Background Soil Data Set. Pesticides/PCBs and polycyclic aromatic hydrocarbons (PAHs) are normally compared to the MCP RCS1 standard when evaluating dredging feasibility. Concentrations of pesticides/PCBs or PAHs did not exceed the MCP RCS1 standard for any of the variables analyzed. There are no thresholds for nutrients, but from experience it is understood that TKN levels of 1000 mg/kg and total phosphorus levels of 100 mg/kg are indicative of nutrient rich sediments, but are not extreme. Based on the sediment analytical results, sediment disposal does not appear likely to pose a regulatory problem.

Analyses for nitrogen and phosphorus in the sediment matrix indicate moderate to high levels (Table 3-3). Sediment nutrient levels are sufficient to support rooted aquatic vegetation. Sediment nutrients may have a significant role in the overall nutrient budget via exchange of nutrients at the sediment/water interface. The extent of sediment/water nutrient exchange is dependent on factors, such as aeration, pH, and other factors. These factors are not well known and are time-variable in the ponds and, as a result, the role of sediment nutrients in the overall nutrient budget of Lake Como is not well known. Clearly, however, nutrients stored in the sediments are supporting rooted aquatic vegetation and act as a source of nutrients to the overlying water column in the Lake Como ponds.

Grain size analyses of sediments from Lake Como indicate that the pond bottoms consist of about 60% sand and 40% silt and clay. The composition in the upstream and downstream lakes was comparable. It should be noted that the terms sand, silt and clay as applied to sediments refer mainly to size fractions and are not the qualitative composition. Much of the sediment is of organic origin (65% and 36%, Table 3-3). The percent of water in sediment samples ranged from 85 to 92%.

Grain size and related physical analyses aid in estimating the time and assist interpretation of sediment quality results. The sediment in Lake Como is typical pond muck, with a high organic content and low overall solids content. Thoroughly dried, the sediment would occupy no more than 25% of the in-place volume, but getting to that level of dryness would be slow and difficult with the observed fine-grained organic matter content. Many contaminants, especially metals, are strongly bound by that organic matrix.

3.5 Biological Survey Results

3.5.1 Phytoplankton

Water samples were analyzed for phytoplankton assemblage composition, density and relative abundance. With the onset of colder autumn weather, diatoms and cryptophytes predictably dominated the assemblage in September 2000 (Table 3-4). Some green and blue-green algae were present, but there was no sign of a typical summer bloom assemblage. Overall cell densities and biomasses were moderate. The assemblage was fairly rich, with 24-25 genera encountered, but was only moderately diverse; a few taxa contributed disproportionately larger numbers of cells to the total. The large numbers of cryptophytes suggest a large quantity of available dissolved organic carbon, typical of autumn conditions in eutrophic lakes.

Samples from May 2001 contained moderate biomasses of algae, with pyrrhophytes (dinoflagellates) dominating the West Pond and cryptophytes again abundant in the Main Pond (Table 3-5). While green algae were numerically abundant, and diatoms, golden algae and cryptophytes were present, no other groups provided substantial biomass. There were fewer taxa than in September, and diversity and evenness were somewhat higher. As with the September samples, there is a strong indication of dependence on organic carbon; the dominant algae are facultative heterotrophs (they can utilize available food sources in addition to photosynthesizing).

The two sets of samples collected in August exhibited distinctly higher biomasses, almost entirely as a function of high dinoflagellate abundance (Tables 3-6 and 3-7). Other algal groups were represented, but only at low densities. Species richness was relatively low, but diversity and evenness were not as depressed as the biomass measures would suggest; dinoflagellates are very large relative to most other algae, so diversity measures based on cell count will not reflect the overwhelming nature of the dinoflagellate biomass. Blue-greens were minor parts of the algal assemblage. Again, strong indication of reliance on organic compounds, in addition to traditional photosynthetic food projection, was found.

Overall, the composition of the phytoplankton suggests high nutrient levels, but a substantial reliance on heterotrophy as well as photosynthesis. High dissolved organic carbon levels are indicated. There is no apparent shortage of nitrogen, based on a lack of nitrogen-fixing blue-green forms. The types of algae present are typically associated with eutrophic conditions where organic inputs are higher than inorganic loads. This could suggest some form of sewage influence, but it is equally plausible that the accumulated organic sediment is controlling algal composition during times of low inflow through sediment-water interactions. August algal abundance was high with August 1 measurements about 34,000 and 22,000 μg/L in the West and Main Ponds, respectively. Levels in excess of 10,000 μg/L level are usually considered to represent bloom condition.

3.5.2 Zooplankton

Zooplankton were assessed only in the Main Pond and only in 2001, and few zooplankton were encountered (Tables 3-8 and 3-9). Density, as individuals or biomass per liter, was very low in all samples, and individual body length was low as well. Biomass was 10 to 30 μg/L, well below the threshold of about 100 μg/L necessary to produce any significant grazing pressure. Average body length was on the order of 0.3 to 0.4 mm, suggesting limited feeding capacity for each individual zooplankter. There were very few individuals with body lengths in excess of 1.0 mm. Rotifers, copepods and cladocerans were observed. Intense fish predation could explain the observed pattern, but it is equally likely that short hydraulic detention time limits development of a more dense zooplankton assemblage. The small size and abundance of zooplankton indicates minimal grazing impact on algae (especially the large dinoflagellates) and a poor food base for fish.

3.5.3 Chlorophyll a

Chlorophyll is a green plant pigment essential to photosynthesis. Measuring the concentration of chlorophyll a in a water sample is a useful indicator of a waterbody's trophic state or degree of nutrient enrichment. Chlorophyll a measurements collected on September 7, 2000 in Lake Como ranged from 0.2 to 9.4 µg/L (Table 3-10). The West Pond had the highest concentration of chlorophyll a (9.4 µg/l). Concentration in the Main Pond was moderate (3.3 µg/L). In general, values exceeding 10 µg/L are characteristic of eutrophic conditions, although some classification systems consider this threshold as low as 4 µg/L. Chlorophyll measurements were not collected during the August survey, but chlorophyll was likely higher during August than September, based on phytoplankton density measurements collected during August and September.

3.5.4 Aquatic Plant Survey

The aquatic plant survey was completed to quantify 1) the amount of biomass in the water column, 2) the amount of coverage on the surface, and 3) the species types present in the water column and on the surface. At the time of the investigation aquatic vegetation in the lake was extremely dense. Much of the water column was occupied by rooted aquatic vegetation and almost the entire surface was covered with a variety of species of filamentous green algae.

The biovolume varied among ponds. However, the distribution and range of biovolume percentages was similar between the ponds (Table 3-11, Figure 3-4). The biovolume percentages were generally highest where the sediment thickness was greatest. This may indicate a better substrate for the growth of rooted aquatic vegetation but is not conclusive. The biovolume percentage in the main pond was generally the lowest along the medial axis of the lake where much of the water transport occurs. This suggests that either the coarser substrate here is not as suitable, or that such vegetation does not grow as well in moving water.

The cover of aquatic vegetation was much more uniform over each of the ponds (Table 3-11, Figure 3-5). During the time of the investigation almost all of Lake Como was covered with a thick mat of algae. The thick coverage occurred over all parts of the lake except for a small area near the outlet of the Main Pond and near the south shore of the Main Pond. There was no obvious reason for the lack of coverage at these locations but they did not detract from the dense growth of filamentous green algae over the rest of the lake.

The Main Pond was heavily dominated by rooted aquatic macrophytes. In most areas, macrophyte densities were significant enough to colonize the entire water column, even at the deepest depths. Fanwort (*Cabomba caroliniana*) was the most abundant species. Fanwort is an aggressive non-native species that may achieve densities great enough to impede recreational activities. Fanwort was not found in the West Pond. Yellow lilies and pond lilies (*Nuphar variegata* and *Nymphaea tuberosa*, respectively) were found along the shoreline. Densities were the greatest in the western portion of the impoundment. Bladderwort (*Utricularia vulgaris*), waterweed (*Elodea canadensis*) and a native milfoil (*Myriophyllum humile*) were observed in the widest portion of the basin. These macrophytes are submersed native species that have the potential to grow to nuisance levels. Filamentous green algae, watermeal (*Wolffia columbiana*), duckweed (*Lemna minor*) covered much of the water surface.

Watermeal and duckweed are native free-floating macrophytes typically found growing together forming dense mats. These mats are often significant enough to shade submersed macrophytes.

The West Pond was dominated by lilies and contained species not observed in the Main Pond. Dense growths of a macroscopic green alga *Nitella flexilis* were observed in the western most portion of the impoundment with sporadic growths throughout the remaining basin. Pondweeds (*Najas flexilis* and *Potamogeton pusillus*) were present in low-moderate densities. Emergent wetland species *Pontederia cordata* (pickerelweed) and *Sagittaria latifolia* (arrowhead) were present along the shoreline in moderate densities.

3.5.5 Wildlife and Fish Observations

Wildlife observed at Lake Como during the field investigation included waterfowl, reptiles, and amphibians. Waterfowl consisted of Canada geese (*Branta canadensis*) and Mute swans (*Cygnus olor*). Reptiles and amphibians included painted turtles (*Chrysemys picta*), American bullfrogs (*Rana catesbeiana*), and Green frogs (*Rana clamitans*).

Fish were not observed during the investigation. A survey specifically designed to enumerate the fish community was beyond the scope of this investigation. However, the Massachusetts Division of Fish and Wildlife (MassWildlife), Southeast Wildlife District, was contacted to ascertain if any fish or wildlife survey had been conducted on Lake Como. According to Steve Hurley at MassWildlife, the agency does not conduct fish or wildlife surveys on small waterbodies, therefore, the Southeast Wildlife District office did not have any information on Lake Como. Mr. Hurley did list some common warm water fish that are likely to inhabit similar ecosystems. It appears unlikely, however, that a diverse and copious fish community is present in Lake Como due to the low summer dissolved oxygen concentrations, low water levels, and minimal food resources (e.g., lack of zooplankton) observed.

3.6 Summary of Field Program Results

All observations obtained during the field program are consistent with characterization of Lake Como as a small, highly eutrophic pond system. Lake Como presently receives too much of a nutrient load and has too little water volume to maintain healthy water quality conditions. Lake Como field program results are summarized briefly below and are categorized as physical, water quality, and sediment/biological observations.

3.6.1 Physical Observations

- Source Waters – 4 storm drains, 1 tributary creek, and groundwater baseflow were identified as source waters for the Lake Como ponds.

- Leak in Outlet Dam – A continuous leak was identified in the Main Pond's outlet dam effectively reducing the water level and storage capacity in the Lake Como system.
- Shallow Pond Bathymetry – The West and Main Ponds are typically 1 to 2 feet deep with relatively small areas as deep as 4 feet.
- Pond Sediment Thickness – A 1 to 2 foot layer of soft, organic sediments was observed in West and Main Ponds.
- Overland Flow for Storm Drainage – 3 storm drains (Attleboro, North Attleboro, and Hospital, as shown in Figure 3-1) are not directly connected to Lake Como surface waters. As a result, water from these storm drains may not reach the ponds except during major precipitation events.

3.6.2 Water Quality Observations

- Low In-Lake Dissolved Oxygen Concentrations – DO concentrations in the two ponds were frequently below the water quality standard of 5.0 mg/L, with measurements as low as 1.0 mg/L obtained. Low DO concentrations are due to the presence of excessive growths of aquatic vegetation in the ponds.
- Excessive In-Lake Nutrient Concentrations - Phosphorus and nitrogen concentrations in the two ponds were observed to be excessive during dry-weather surveys and were sufficient to support eutrophic conditions. For example, in-lake phosphorus concentrations ranged from 30 to 310 µg/L, with a mean value of 90 µg/L.
- Excessive Stormwater Nutrient Concentrations – Phosphorus and nitrogen concentrations from the 4 storm drains were observed to be excessive during wet-weather surveys and appear to be a major source of excessive in-lake nutrient concentrations. For example, phosphorus stormwater concentrations ranged from 60 to 6,200 µg/L, with a mean value of 1,760 µg/L.

3.6.3 Sediment and Biological Observations

- Lack of Pond Sediment Toxicity - Pond sediments were found not to have elevated levels of metals or other toxins, indicating that dredging would be relatively straightforward from a dredged material handling perspective.
- Sediment Nutrients – Nutrient levels in sediments were moderate to high indicating that sediments may act as a significant source in the overall budget, depending on oxygen levels, pH, and other factors.

- Extensive Aquatic Biological Growth - Extensive growth of phytoplankton and rooted aquatic vegetation was observed during the summertime survey. The water surface was covered with floating macrophytes diminishing the potential for recreational uses.

An evaluation of the Lake Como watershed is provided in Section 4, followed by a Lake Como watershed modeling application and a set of lake restoration recommendations, in Sections 5 and 6.

Table 3-1 Water Quality Results: Dry-Weather Sampling

Station Name	Nitrate-N (mg/L)	Ammonium-N (mg/L)	TKN (mg/L)	Total Phosphorus (mg/L)	Dissolved Phosphorus (mg/L)	Fecal Coliform (/100ml)	Fecal Streptoccus (/100 ml)	Temp. (°C)	Dissolved Oxygen (mg/L)	pH (S.U.)	Alkalinity (mg/L)	Chloride (mg/L)	Specific Conductivity (uS/cm)	Turbidity (NTU)	TDS (mg/L)	TSS (mg/L)	Flow (cfs)
September 7, 2001																	
Main pond	<0.01	0.01	2.7	0.29	--	160	--	17.7	1.4	7.1	67	22.7	203	19	134	96	NA
West Pond	<0.01	0.01	1.5	0.14	--	18	--	20.1	5.6	7.2	70	24.4	229	4.5	159	24	NA
Inlet	0.01	0.02	0.82	0.07	--	--	--	20.9	9.7	8.4	44	19.0	155	1.5	145	34	0
Outlet	<0.01	0.01	0.27	0.03	--	--	--	22.3	4.8	7.5	64	22.9	211	2.2	134	6	0.00
May 29, 2001																	
Main Pond	<0.01	0.01	1.7	0.06	0.04	480	10	20	7.9	6.9	46	35.6	225	3.4	NA		
West Pond	<0.01	0.03	1.1	0.08	0.05	270	40	21.9	4.5	6.6	41	26.9	188	2.6	0.58		
Inlet	<0.01	<0.01	0.9	0.05	0.04	--	--	16.3	3.7	6.4	39	21.9	168	2.6	0.36		
Outlet	0.12	0.04	1.0	0.06	0.04	--	--	19.3	7.6	6.9	53	57.3	314	4.0	1.17		
Esker Village	1.62	0.01	0.3	0.08	0.04	--	--	14.8	10.1	7.9	77	50.3	346	5.2	<0.1		

Table 3-1 Water Quality Results: Dry-Weather Sampling (continued)

Station Name	Nitrate-N (mg/L)	Ammonium-N (mg/L)	TKN (mg/L)	Total Phosphorus (mg/L)	Dissolved Phosphorus (mg/L)	Fecal Coliform (/100ml)	Fecal Streptoccus (/100 ml)	Temp. (°C)	Dissolved Oxygen (mg/L)	pH (S.U.)	Alkalinity (mg/L)	Chloride (mg/L)	Specific Conductivity (uS/cm)	Turbidity (NTU)	TDS (mg/L)	TSS (mg/L)	Flow (cfs)
August 1, 2001																	
Main Pond	<0.01	0.02	0.8	0.06	0.04	620	10	27.4	3.0	7.0	52	37.6	221	2.6	NA		
West Pond	<0.02	0.04	1.4	0.12	0.03	940	10	25.2	7.1	7.2	64	43.9	280	4.5	<0.1		
Inlet	<0.04	0.08	4.0	0.31	0.04	--	--	19.7	1.0	7.0	66	33.8	244	24.0	0		
Outlet	<0.06	0.12	0.9	0.08	0.05	--	--	22.6	3.0	7.0	64	41.6	276	5.0	<0.1		
August 30, 2001																	
Main Pond	<0.01	0.03	0.9	0.04	0.02	2400	70	26.7	5.5	7.0	46	26.9	198	2.0	--		
West Pond	<0.01	<0.01	0.8	0.03	0.02	1800	30	25	4.6	7.0	52	33.3	234	1.3	<1		
Inlet	<0.01	0.02	2.1	0.15	0.03	--	--	19.5	1.6	6.5	46	26.2	202	5.4	0		
Outlet	0.03	0.1	0.9	0.06	0.02	--	--	22.4	4.2	6.8	60	39.4	272	2.7	<1		
Statistical Summary																	
Minimum	<0.01	<0.01	0.3	0.03	0.02	18	10	14.8	1.0	6.4	39	19.0	155	1.3			
Maximum	1.62	0.12	4.0	0.31	0.05	1800	40	25.2	10.1	8.4	77	57.3	346	24.0			
Mean *	0.11	0.03	1.1	0.09	0.04	836	28	19.3	4.7	7.1	55	33.1	234	5.4			

* one half the detection limit was used for values below detection (ex. 0.005 was used for 0.01)

Table 3-2 Water Quality Results: Wet Weather Sampling

Station Name	Nitrate-N (mg/L)	Ammonium-N (mg/L)	TKN (mg/L)	Total Phosphorus (mg/L)	Alkalinity (mg/L)	Chloride (mg/L)	Turbidity (NTU)	TDS (mg/L)	TSS (mg/L)
September 13, 2000									
North Attleboro	2.44	1.01	2.8	0.63	8	5.9	36	126	108
Esker Village	2.30	0.63	1.8	0.33	38	10.5	29	134	80
June 18, 2001									
Attleboro	1.61	0.10	4.4	1.6	40	5.8	64	111	468
Fuller Hospital	2.14	0.23	7.3	6.2	56	36.3	55	203	660
June 27, 2002									
Esker Village	1.2	0.69	0.71	0.06	-----	-----	-----	218	7
Statistical Summary									
Minimum	1.2	0.10	0.71	0.06	8	5.8	29	111	7
Maximum	2.44	1.01	7.3	6.20	56	36.3	64	218	660
Mean *	1.94	0.53	3.4	1.76	36	14.6	46	158	265

* one half the detection limit was used for values below detection (ex. 0.005 was used for 0.01)

Table 3-3 Phytoplankton Density of Samples Collected in the West and Main Ponds: September 7, 2000

	PHYTOPLANKTON DENSITY			
	(Cells/ML)		(UG/L)	
	West Pond	Main Pond	West Pond	Main Pond
TAXON	9/7/2000	9/7/2000	9/7/2000	9/7/2000
BACILLARIOPHYTA				
Achnanthes	28	22	2.8	2.2
Cocconeis	14	22	5.6	8.8
Cyclotella	7	0	0.7	0.0
Cymbella	35	0	35.0	0.0
Epithemia	7	0	33.6	0.0
Eunotia	56	44	140.0	110.0
Fragilaria	140	550	42.0	165.0
Gomphonema	112	110	112.0	158.4
Melosira	56	44	16.8	13.2
Navicula	182	44	91.0	71.5
Nitzschia	182	110	145.6	147.4
Stauroneis	7	0	280.0	0.0
Synedra	56	88	246.4	387.2
Tabellaria	14	0	11.2	0.0
CHLOROPHYTA				
Actinastrum	21	22	2.1	2.2
Closterium	7	11	700.0	44.0
Coelastrum	0	88	0.0	17.6
Cosmarium	7	11	56.0	8.8
Pediastrum	56	44	11.2	8.8
Scenedesmus	0	44	0.0	4.4
Staurodesmus	7	0	70.0	0.0
Tetraedron	0	11	0.0	6.6
CHRYSOPHYTA				
Dinobryon	56	0	168.0	0.0
Synura	357	33	285.6	26.4
CRYPTOPHYTA				
Cryptomonas	308	2376	718.2	5825.6
Rhodomonas	28	154	5.6	30.8
CYANOPHYTA				
Anabaena	280	110	56.0	22.0
Oscillatoria	840	440	8.4	4.4
EUGLENOPHYTA				
Euglena	0	22	0.0	11.0
Phacus	0	11	0.0	3.3
Trachelomonas	0	22	0.0	22.0

Table 3-3 Phytoplankton Density of Samples Collected in the West and Main Ponds: September 7, 2000 (continued)

	PHYTOPLANKTON DENSITY			
	(Cells/ML)		(UG/L)	
	West Pond	Main Pond	West Pond	Main Pond
TAXON	9/7/2000	9/7/2000	9/7/2000	9/7/2000
Pyrrhophyta				
Rhodophyta				
SUMMARY STATISTICS				
DENSITY (#/ML)				
Bacillariophyta	896	1034	1162.7	1063.7
Chlorophyta	98	231	839.3	92.4
Chrysophyta	413	33	453.6	26.4
Cryptophyta	336	2530	723.8	5856.4
Cyanophyta	1120	550	64.4	26.4
Euglenophyta	0	55	0.0	36.3
Pyrrhophyta	0	0	0.0	0.0
Rhodophyta	0	0	0.0	0.0
Total Phytoplankton	2863	4433	3243.8	7101.6
TAXONOMIC RICHNESS				
Bacillariophyta	14	9		
Chlorophyta	5	7		
Chrysophyta	2	1		
Cryptophyta	2	2		
Cyanophyta	2	2		
Euglenophyta	0	3		
Pyrrhophyta	0	0		
Rhodophyta	0	0		
Total Phytoplankton	25	24		
S-W DIVERSITY INDEX	1.05	0.79		
EVENNESS INDEX	0.75	0.57		

Table 3-4 Phytoplankton Density of Samples Collected in the West and Main Ponds: May 29, 2001

	PHYTOPLANKTON DENSITY			
	(Cells/ML)		(UG/L)	
	West Pond	Main Pond	West Pond	Main Pond
TAXON	5/29/2001	5/29/2001	5/29/2001	5/29/2001
BACILLARIOPHYTA				
Cyclotella	0	38	0.0	95.0
Eunotia	135	0	135.0	0.0
Fragilaria	0	76	0.0	22.8
Gomphonema	0	76	0.0	76.0
Navicula	90	38	45.0	19.0
Nitzschia	90	38	72.0	30.4
Synedra	90	0	72.0	0.0
Tabellaria	0	38	0.0	30.4
CHLOROPHYTA				
Ankistrodesmus	0	456	0.0	45.6
Chlamydomonas	135	38	13.5	3.8
Chlorococcum	0	0	0.0	0.0
Closterium	0	0	0.0	0.0
Coelastrum	360	0	72.0	0.0
Crucigenia	1440	0	144.0	0.0
Dictyosphaerium	180	0	18.0	0.0
Oocystis	0	0	0.0	0.0
Pandorina	0	0	0.0	0.0
Pediastrum	0	0	0.0	0.0
Scenedesmus	90	0	9.0	0.0
Sphaerocystis	0	0	0.0	0.0
Staurastrum	0	38	0.0	30.4
CHRYSOPHYTA				
Dinobryon	0	0	0.0	0.0
Mallomonas	0	0	0.0	0.0
Ochromonas	180	76	18.0	7.6
Synura	0	0	0.0	0.0
CRYPTOPHYTA				
Cryptomonas	90	532	18.0	1109.6
CYANOPHYTA				
Oscillatoria	990	1824	9.9	18.2
EUGLENOPHYTA				
Euglena	0	0	0.0	0.0
Phacus	0	0	0.0	0.0
Trachelomonas	0	0	0.0	0.0
PYRRHOPHYTA				
Ceratium	0	0	0.0	0.0

Table 3-4 Phytoplankton Density of Samples Collected in the West and Main Ponds: May 29, 2001 (continued)

TAXON	PHYTOPLANKTON DENSITY (Cells/ML) West Pond 5/29/2001	(Cells/ML) Main Pond 5/29/2001	(UG/L) West Pond 5/29/2001	(UG/L) Main Pond 5/29/2001
Peridinium	45	0	2025.0	0.0
Rhodophyta				
SUMMARY STATISTICS				
DENSITY (#/ML)				
Bacillariophyta	405	304	324.0	273.6
Chlorophyta	2205	532	256.5	79.8
Chrysophyta	180	76	18.0	7.6
Cryptophyta	90	532	18.0	1109.6
Cyanophyta	990	1824	9.9	18.2
Euglenophyta	0	0	0.0	0.0
Pyrrhophyta	45	0	2025.0	0.0
Rhodophyta	0	0	0.0	0.0
Total Phytoplankton	3915	3268	2651.4	1488.8
TAXONOMIC RICHNESS				
Bacillariophyta	4	6		
Chlorophyta	5	3		
Chrysophyta	1	1		
Cryptophyta	1	1		
Cyanophyta	1	1		
Euglenophyta	0	0		
Pyrrhophyta	1	0		
Rhodophyta	0	0		
Total Phytoplankton	13	12		
S-W DIVERSITY INDEX	0.84	0.64		
EVENNESS INDEX	0.75	0.59		

Table 3-5 Phytoplankton Density of Samples Collected in the West and Main Ponds: August 1, 2001

	PHYTOPLANKTON DENSITY			
	(Cells/ML)		(UG/L)	
	West Pond	Main Pond	West Pond	Main Pond
TAXON	**8/1/2001**	**8/1/2001**	**8/1/2001**	**8/1/2001**
BACILLARIOPHYTA				
Cyclotella	42	0	105.0	0.0
Eunotia	0	0	0.0	0.0
Fragilaria	0	0	0.0	0.0
Gomphonema	42	0	42.0	0.0
Navicula	42	0	21.0	0.0
Nitzschia	42	0	33.6	0.0
Synedra	42	0	33.6	0.0
Tabellaria	0	0	0.0	0.0
CHLOROPHYTA				
Ankistrodesmus	0	42	0.0	4.2
Chlamydomonas	0	0	0.0	0.0
Chlorococcum	0	0	0.0	0.0
Closterium	0	0	0.0	0.0
Coelastrum	0	0	0.0	0.0
Crucigenia	0	0	0.0	0.0
Dictyosphaerium	0	0	0.0	0.0
Oocystis	0	0	0.0	0.0
Pandorina	0	0	0.0	0.0
Pediastrum	672	0	134.4	0.0
Scenedesmus	0	0	0.0	0.0
Sphaerocystis	0	0	0.0	0.0
Staurastrum	0	0	0.0	0.0
CHRYSOPHYTA				
Dinobryon	0	0	0.0	0.0
Mallomonas	0	0	0.0	0.0
Ochromonas	0	0	0.0	0.0
Synura	84	0	67.2	0.0
CRYPTOPHYTA				
Cryptomonas	588	1218	940.8	1125.6
CYANOPHYTA				
Oscillatoria	0	0	0.0	0.0
EUGLENOPHYTA				
Euglena	42	42	21.0	21.0
Phacus	0	42	0.0	12.6
Trachelomonas	42	42	42.0	42.0

	PHYTOPLANKTON DENSITY			
	(Cells/ML)		(UG/L)	
	West Pond	Main Pond	West Pond	Main Pond
TAXON	8/1/2001	8/1/2001	8/1/2001	8/1/2001
PYRRHOPHYTA				
Ceratium	0	0	0.0	0.0

Table 3-5 Phytoplankton Density of Samples Collected in the West and Main Ponds: August 1, 2001 (continued)

	PHYTOPLANKTON DENSITY			
	(Cells/ML)		(UG/L)	
	West Pond	Main Pond	West Pond	Main Pond
TAXON	8/1/2001	8/1/2001	8/1/2001	8/1/2001
Peridinium	714	462	32130.0	20790.0
RHODOPHYTA				
SUMMARY STATISTICS				
DENSITY (#/ML)				
Bacillariophyta	210	0	235.2	0.0
Chlorophyta	672	42	134.4	4.2
Chrysophyta	84	0	67.2	0.0
Cryptophyta	588	1218	940.8	1125.6
Cyanophyta	0	0	0.0	0.0
Euglenophyta	84	126	63.0	75.6
Pyrrhophyta	714	462	32130.0	20790.0
Rhodophyta	0	0	0.0	0.0
Total Phytoplankton	2352	1848	33570.6	21995.4
TAXONOMIC RICHNESS				
Bacillariophyta	5	0		
Chlorophyta	1	1		
Chrysophyta	1	0		
Cryptophyta	1	1		
Cyanophyta	0	0		
Euglenophyta	2	3		
Pyrrhophyta	1	1		
Rhodophyta	0	0		
Total Phytoplankton	11	6		
S-W DIVERSITY INDEX	0.73	0.42		
EVENNESS INDEX	0.70	0.54		

Table 3-6 Phytoplankton Density of Samples Collected in the West and Main Ponds: August 30, 2001

	PHYTOPLANKTON DENSITY			
	(Cells/ML)		(UG/L)	
	West Pond	Main Pond	West Pond	Main Pond
TAXON	8/30/2001	8/30/2001	8/30/2001	8/30/2001
BACILLARIOPHYTA				
Cyclotella	0	0	0.0	0.0
Eunotia	0	0	0.0	0.0
Fragilaria	0	0	0.0	0.0
Gomphonema	0	0	0.0	0.0
Navicula	42	0	21.0	0.0
Nitzschia	84	90	67.2	72.0
Synedra	42	0	33.6	0.0
Tabellaria	0	0	0.0	0.0
CHLOROPHYTA				
Ankistrodesmus	42	0	4.2	0.0
Chlamydomonas	0	0	0.0	0.0
Chlorococcum	0	810	0.0	81.0
Closterium	0	45	0.0	180.0
Coelastrum	0	0	0.0	0.0
Crucigenia	0	0	0.0	0.0
Dictyosphaerium	0	180	0.0	18.0
Oocystis	168	0	67.2	0.0
Pandorina	336	0	134.4	0.0
Pediastrum	0	0	0.0	0.0
Scenedesmus	0	0	0.0	0.0
Sphaerocystis	0	180	0.0	36.0
Staurastrum	0	0	0.0	0.0
CHRYSOPHYTA				
Dinobryon	0	45	0.0	135.0
Mallomonas	0	90	0.0	202.5
Ochromonas	0	0	0.0	0.0
Synura	168	0	134.4	0.0
CRYPTOPHYTA				
Cryptomonas	378	270	369.6	243.0
CYANOPHYTA				
Oscillatoria	1176	0	11.8	0.0
EUGLENOPHYTA				
Euglena	42	0	21.0	0.0
Phacus	0	0	0.0	0.0
Trachelomonas	42	315	42.0	702.0

Table 3-6 Phytoplankton Density of Samples Collected in the West and Main Ponds: August 30, 2001 (continued)

	PHYTOPLANKTON DENSITY			
	(Cells/ML)		(UG/L)	
	West Pond	Main Pond	West Pond	Main Pond
TAXON	**8/30/2001**	**8/30/2001**	**8/30/2001**	**8/30/2001**
PYRRHOPHYTA				
Ceratium	0	90	0.0	8460.0
SUMMARY STATISTICS				
DENSITY (#/ML)				
Bacillariophyta	168	90	121.8	72.0
Chlorophyta	546	1215	205.8	315.0
Chrysophyta	168	135	134.4	337.5
Cryptophyta	378	270	369.6	243.0
Cyanophyta	1176	0	11.8	0.0
Euglenophyta	84	315	63.0	702.0
Pyrrhophyta	336	135	15120.0	10485.0
Rhodophyta	0	0	0.0	0.0
Total Phytoplankton	2856	2160	16026.4	12154.5
TAXONOMIC RICHNESS				
Bacillariophyta	3	1		
Chlorophyta	3	4		
Chrysophyta	1	2		
Cryptophyta	1	1		
Cyanophyta	1	0		
Euglenophyta	2	1		
Pyrrhophyta	1	2		
Rhodophyta	0	0		
Total Phytoplankton	12	11		
S-W DIVERSITY INDEX	0.82	0.85		
EVENNESS INDEX	0.76	0.82		

Table 3-7 Summary Statistics of Zooplankton: May 29, 2001

TAXON	ZOOPLANKTON DENSITY (#/L) L. Como Main 5/29/2001	ZOOPLANKTON BIOMASS (UG/L) L. Como Main 5/29/2001
PROTOZOA		
Ciliophora	0.0	0.0
Mastigophora	0.0	0.0
Sarcodina	0.0	0.0
ROTIFERA		
Asplanchna	0.0	0.0
Brachionus	0.3	0.0
Kellicottia	0.0	0.0
Keratella	0.6	0.1
Polyarthra	0.0	0.0
COPEPODA		
Copepoda-Cyclopoida		
Cyclops	1.9	4.7
Mesocyclops	0.0	0.0
Copepoda-Calanoida		
Copepoda-Harpacticoida	0.0	0.0
Other Copepoda-Adults	0.0	0.0
Other Copepoda-Copepodites	0.0	0.0
Other Copepoda-Nauplii	3.2	8.5
CLADOCERA		
Bosmina	0.0	0.0
Ceriodaphnia	1.1	2.7
Chydorus	0.6	0.6
Daphnia ambigua	0.0	0.0
OTHER ZOOPLANKTON		
SUMMARY STATISTICS		
PROTOZOA	0.0	0.0
ROTIFERA	1.0	0.1
COPEPODA	5.1	13.2
CLADOCERA	1.7	3.4
OTHER ZOOPLANKTON	0.0	0.0
TOTAL ZOOPLANKTON	7.8	16.6
TAXONOMIC RICHNESS		
PROTOZOA	0	
ROTIFERA	2	
COPEPODA	2	
CLADOCERA	2	
OTHER ZOOPLANKTON	0	
TOTAL ZOOPLANKTON	6	
S-W DIVERSITY INDEX	0.66	
EVENNESS INDEX	0.85	
MEAN LENGTH: ALL FORMS	0.40	
MEAN LENGTH: CRUSTACEANS	0.44	

Table 3-8 Summary Statistics of Zooplankton: August 1, 2001

TAXON	ZOOPLANKTON DENSITY (#/L) L. Como Main 8/1/2001	ZOOPLANKTON BIOMASS (UG/L) L. Como Main 8/1/2001
PROTOZOA		
Ciliophora	0.0	0.0
Mastigophora	0.0	0.0
Sarcodina	0.0	0.0
ROTIFERA		
Asplanchna	0.0	0.0
Brachionus	0.0	0.0
Kellicottia	2.0	0.1
Keratella	0.4	0.0
Polyarthra	0.0	0.0
COPEPODA		
Copepoda-Cyclopoida		
Cyclops	0.0	0.0
Mesocyclops	0.4	0.5
Copepoda-Calanoida		
Copepoda-Harpacticoida	0.0	0.0
Other Copepoda-Adults	0.0	0.0
Other Copepoda-Copepodites	0.0	0.0
Other Copepoda-Nauplii	0.8	2.1
CLADOCERA		
Bosmina	7.0	6.8
Ceriodaphnia	0.0	0.0
Chydorus	0.0	0.0
Daphnia ambigua	0.1	0.2
OTHER ZOOPLANKTON		
SUMMARY STATISTICS		
PROTOZOA	0.0	0.0
ROTIFERA	2.4	0.1
COPEPODA	1.2	2.6
CLADOCERA	7.1	7.0
OTHER ZOOPLANKTON	0.0	0.0
TOTAL ZOOPLANKTON	10.7	9.8
TAXONOMIC RICHNESS		
PROTOZOA	0	
ROTIFERA	2	
COPEPODA	2	
CLADOCERA	2	
OTHER ZOOPLANKTON	0	
TOTAL ZOOPLANKTON	6	
S-W DIVERSITY INDEX	0.47	
EVENNESS INDEX	0.60	
MEAN LENGTH: ALL FORMS	0.27	
MEAN LENGTH: CRUSTACEANS	0.31	

Table 3-9 Chlorophyll a Concentrations Measured in the Lake Como System

Sample ID	Total Chl a (ug/l) Sept 7, 2000	Total Chl a (ug/l) May 29, 2001	Total Chl a (ug/l) Aug 1, 2001	Total Chl a (ug/l) Aug 30, 2001
West Pond	0.19	0.42	6.31	3.06
Main Pond	3.31	0.23	5.15	7.58
Lake Outlet	3.59	N/A	N/A	N/A
Inlet	9.43	N/A	142.91	12.83

N/A not sampled

Table 3-10 Sediment Chemistry Results for Sediments in Lake Como

		MA Mean Lake and Pond Sediment Data	MA DEP Background Soil Data Set 90th Percentile	MCP RCS1	Upstream Lake Como	Downstream Lake Como
Nutrients (mg/kg)						
Kjeldahl Nitrogen	417E, SM 16th Ed.				1140	1210
Phosphorus	4500-P-E-SM 18th Ed.				58	180
Total Metals (mg/kg)						
Arsenic	6010B, SW-846	17.1	16.7	30	<7.18	8.18
Cadmium	6010B, SW-846	4.6	2.06	30	<2.87	<1.36
Chromium	6010B, SW-846	23	28.6	1000	8.80	15.1
Copper	6010B, SW-846	41.8	37.7	1000	28.2	32.1
Lead	6010B, SW-846	203	98.7	300	42.8	70.7
Mercury	7471, EPA 1986		0.28		0.24	0.27
Nickel			16.6		<28.7	<13.6
Vanadium			28.5		<28.0	17
Zinc	6010, EPA 1987	195		2500	69.0	119
PCBs/Pesticides (ug/kg)						
Aldrin	EPA 8080				<0.02	<0.02
alpha-BHC	EPA 8080				<0.015	<0.015
beta-BHC	EPA 8080				<0.03	<0.03
delta-BHC	EPA 8080				<0.045	<0.045
gamma BHC (Lindane)	EPA 8080				<0.02	<0.02
Chlordane	EPA 8080				<0.35	<0.35
4,4'-DDD	EPA 8080				<0.055	<0.055
4,4'-DDE	EPA 8080				<0.02	<0.02
4,4'-DDT	EPA 8080				<0.06	<0.06
Dieldrin	EPA 8080				<0.01	<0.01
Endosulfan I	EPA 8080				<0.07	<0.07
Endosulfan II	EPA 8080				<0.02	<0.02
Endosulfan Sulfate	EPA 8080				<0.33	<0.33
Endrin	EPA 8080				<0.03	<0.03
Endrin Aldehyde	EPA 8080				<0.115	<0.115
Heptachlor	EPA 8080				<0.015	<0.015
Heptachlor Epoxide	EPA 8080				<0.415	<0.415
Methoxychlor	EPA 8080				<0.06	<0.06
Toxaphene	EPA 8080				<1.2	<1.2
PCB-1016	EPA 8080				<100	<100
PCB-1221	EPA 8080				<100	<100
PCB-1232	EPA 8080				<100	<100
PCB-1242	EPA 8080				<100	<100
PCB-1248	EPA 8080				<100	<100
PCB-1254	EPA 8080				<100	<100
PCB-1260	EPA 8080				<100	<100
PCB-1268	EPA 8080				<100	<100
TCMX (Surrogate)					69.9%	75.6%
DCB (Surrogate)					71.9%	80.4%
Petroleum Hydrocarbons	418.2, EPA 1983				95.0%	31.8%

Table 3-10 Sediment Chemistry Results for Sediments in Lake Como

		MA Mean Lake and Pond Sediment Data	MA DEP Background Soil Data Set 90th Percentile	MCP RCS1	Upstream Lake Como	Downstream Lake Como
Polynuclear Aromatic Hyrdocarbons (mg/kg)						
Acenaphthene	EPA 8270			20	<2.270	<1.200
Acenaphthylene	EPA 8270			100	<2.270	<1.200
Anthracene	EPA 8270			1000	<2.270	<1.200
Benzo(a)anthracene	EPA 8270			0.7	<1.140	<0.568
Benzo(b)fluoranthene	EPA 8270			0.7	<0.568	<2.990
Benzo(k)fluoranthene	EPA 8270			7	<0.568	<2.990
Benzo(a)pyrene	EPA 8270			0.7	<1.140	<0.568
Benzo(g,h,i)perylene	EPA 8270			100	<1.140	<0.568
Chrysene	EPA 8270			7	<1.140	<0.568
Dibenzo(a,h)anthracene	EPA 8270			0.7	<1.140	<0.568
Fluoranthene	EPA 8270			1000	<1.140	<0.568
Fluorene	EPA 8270			400	<3.410	<1.790
Indeno(1,2,3-cd)pyrene	EPA 8270			0.7	<1.140	<0.568
Naphthalene	EPA 8270			4	<2.270	<1.200
2-Methyl Naphthalene	EPA 8270				<1.140	<0.568
Phenanthrene	EPA 8270			100	<1.140	<0.568
Pyrene	EPA 8270			700	<3.410	<1.790
2-Fluorobiphenyl (Surrogate)					47.0%	35.8%
Nitrobenzene-D5 (Surrogate)					46.3%	35.4%
Terphenyl-D14 (Surrogate)					65.5%	58.1%
Solids						
Total Organic Carbon	EPA 415.1				8.09%	8.01%
Total Solids (ppm)	2540B SM 18th, 1992				7.77%	14.5%
Total Volatile solids (ppm)	2540G SM 18th, 1992				65.0%	36.1%
Percent Solids					8.3%	14.8%
Grain Size						
% finer than 4.75 mm (Sieve Size 4)					100.0%	100.0%
% finer than 2.00 mm (Sieve Size 10)					98.5%	99.5%
% finer than 0.850 mm (Sieve Size 20)					92.5%	96.3%
% finer than 0.425 mm (Sieve Size 40)					75.5%	81.1%
% finer than 0.300 mm (Sieve Size 50)					67.0%	74.8%
% finer than 0.180 mm (Sieve Size 180)					53.6%	57.9%
% finer than 0.075 mm (Sieve Size 200)					35.4%	43.2%
% > 3"					0.0%	0.0%
% Gravel					0.0%	0.0%
% Sand					64.6%	56.8%
% Silt & Clay					35.4%	43.2%

Table 3-11 Summary of Aquatic Vegetation Investigation in Lake Como

Upstream Impoundment

Transect Point	% Cover (see Legend)	% Biovolume (see Legend)	Species Composition (%)
A-1	4	2	Wc(70), FG(30)
A-2	4	2	Nv(30), Nitella flexilis(20), Najas flexilis(20), FG(20), Wc(10)
A-3	4	3	Nitella flexilis(70), Nt(10), FG(10), Wc(10)
A-4	4	2	Pp(60), FG(20), Nv(10), Wc(10)
B-1	4	2	Nv(60), FG(30), Wc(10)
B-2	4	1	Nv(50), Pp(20), FG(20), Wc(10)
B-3	4	2	Nv(25), Pp(25), Nitella flexilis(25), FG(15), Wc(10)
B-4	3	1	Nitella flexilis(50), Pc(10), Sl(10), Pp(10), Nv(10), FG(5), Wc(5),
C-1	4	1	Pc(60), Nitella flexilis(20), Nv(10), FG(5), Wc(5), Lm(<5)
C-2	4	2	Nv(30), Pp(30), FG(20), Nt(10), Wc(10), Lm(<5)
C-3	4	2	Nv(70), FG(15), Wc(10), Lm(5)
D-1	4	2	Nv(80), Wc(10), FG(10), Lm(<5)
D-2	4	2	Nv(60), FG(20), Wc(10), Pp(5), Lm(5)
E-1	4	1	Sl(40), Pc(30), Nv(10), FG(10), Wc(5), Lm(5)
F-1	4	1	FG(50), Wc(50), Lm(<5)
F-2	4	2	Nv(80), FG(10), Lm(10), Wc(<5)
G-1	4	3	Pp(80), FG(10), Wc(10), Lm(<5)
G-2	4	2	Nv(80), Wc(10), FG(10), Lm(<5)
G-3	4	2	Nv(75), Wc(15), FG(10), Lm(<5)
H-1	4	4	Nitella flexilis(70), Nv(10), Wc(10), FG(10), Lm(<5)
H-2	4	4	Nitella flexilis(70), Nv(10), Wc(10), FG(10), Lm(<5)
I-1	4	3	Nitella flexilis(40), Pp(40), Nv(10), FG(5), Wc(5), Lm(<5)
I-2	4	4	Nitella flexilis(60), FG(20), Nv(10), Wc(5), Lm(5)

Downstream Impondment

Transect Point	% Cover (see Legend)	% Biovolume (see Legend)	Species Composition (%)
A-1	4	2	Cc(60), Ec(30), Wc(10)
A-2	4	3	Cc(60), FG(30), Wc(10)
A-3	4	1	Wc(50), Cc(50)
A-4	4	1	Cc(50), Wc(25), FG(25)
A-5	4	4	Cc(50), FG(40), Wc(10)
B-1	4	4	Cc(60), Nt(30), Wc(5), FG(5)
B-2	4	4	Cc(50), Us(40), Wc(5),FG(5)
B-3	1	1	Cc(100)
B-4	1	1	Cc(100)
B-5	4	4	Cc(30), Nt(30), FG(30),Wc(10)
C-1	4	2	Nt(60), FG(30), Wc(10)
C-2	4	1	Nt(50), FG(40), Wc(10)
C-3	4	4	Cc(60), FG(30), Wc(10)
C-4	3	2	Cc(50), Us(30),FG(10),Wc(6),Lm(4)
C-5	4	2	Nt(80), Wc(10), Cc(10)
D-1	4	3	Nt(20),Cc(20), Ec(20), Mh(20), FG(10), Wc(6), Lm(4)
D-2	4	4	Cc(50), FG(20), Mh(20), Wc(10)
D-3	4	4	Mh(60), Cc(20), FG(10), Wc(10)
D-4	4	4	Cc(80), FG(10), Wc(10)
D-5	1	1	Cc(90), Wc(10)
D-6	2	1	Nt(50), FG(40), Wc(10)
E-1	4	1	Cc(40), FG(40), Wc(20)
E-2	4	4	Nv(25), Cc(25), Mh(25), FG(15), Wc(10)
E-3	4	4	Nv(25), Cc(25), Mh(25), FG(15), Wc(10)
F-1	4	1	Wc(45), FG(45), Lm(10)
G-1	4	1	FG(50), Wc(50)

Photo #1:
North Attleboro Storm Drain.
Downstream of Como Drive.

Photo #2:
Attleboro Storm Drain.
Downstream of Como Drive.

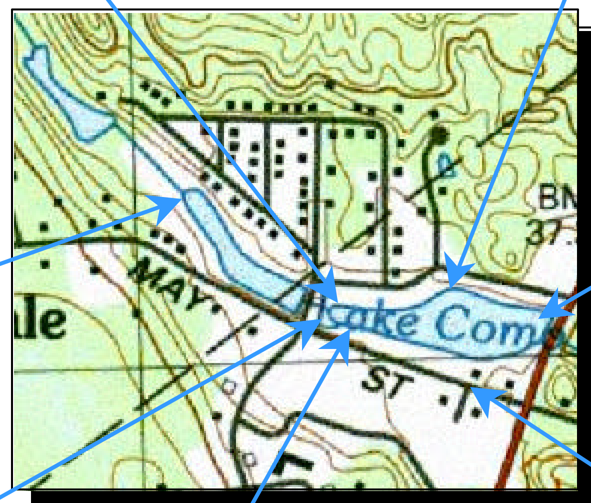

Photo #3:
Main Basin Outlet.
Upstream of Route 1.

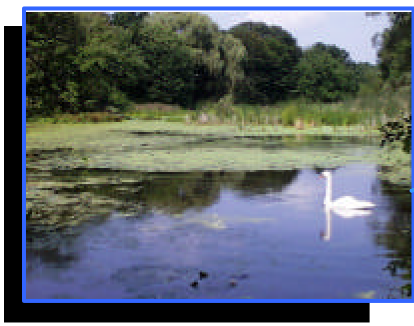

Photo #7:
West Basin
looking Upstream.

Photo #4:
Fuller Hospital
Storm Drain.
Downstream
of May Street.

Photo #6:
Main Basin Inlet.
Downstream of
Heather Street.

Photo #5:
Esker Village
Storm Drain.
Downstream of
May Street.

Map of Lake Como with Photographs of Key Features

Figure 3-1

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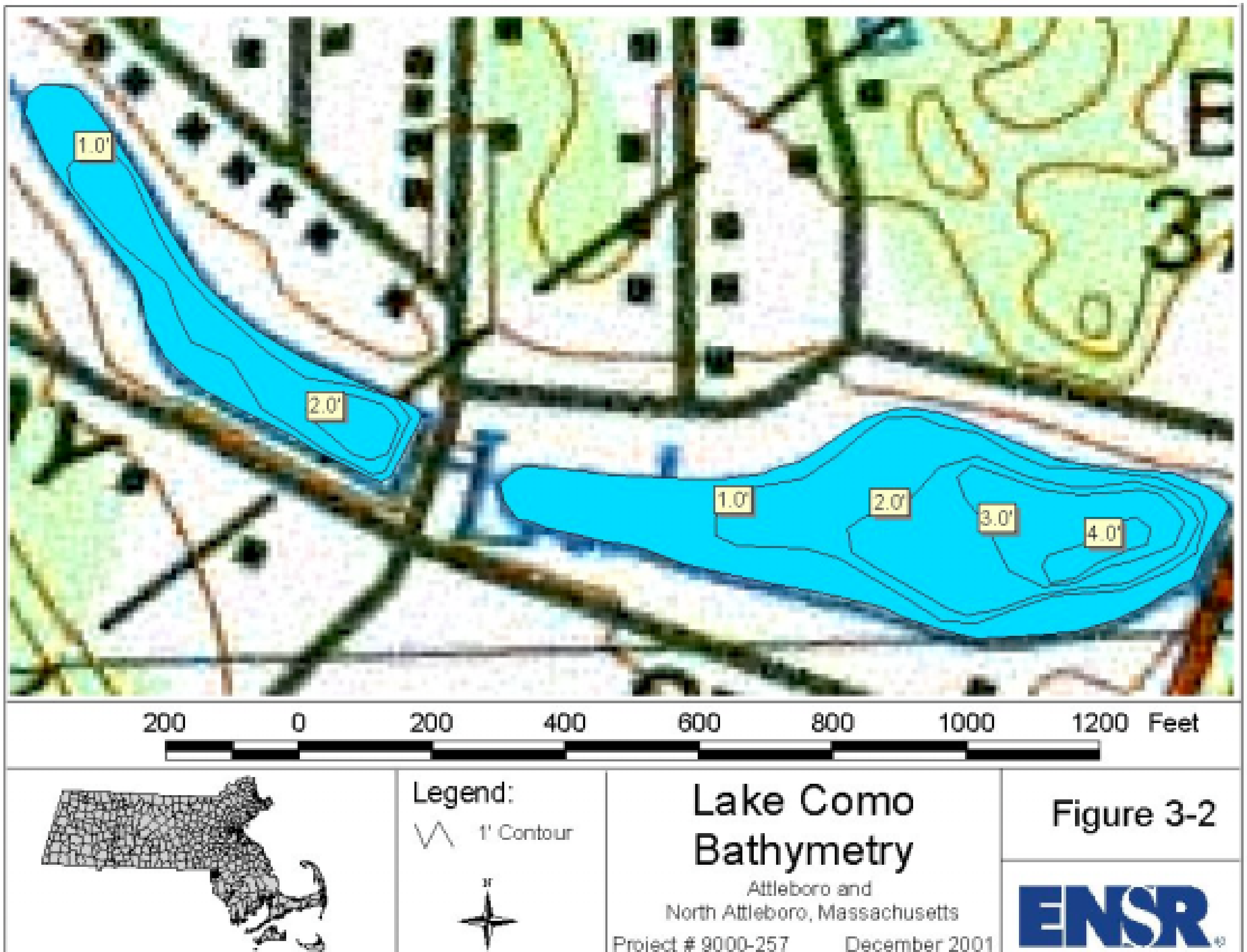
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2.0'
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3.0'
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200 0 200 400 600 800 1000 1200 Feet
Legend:
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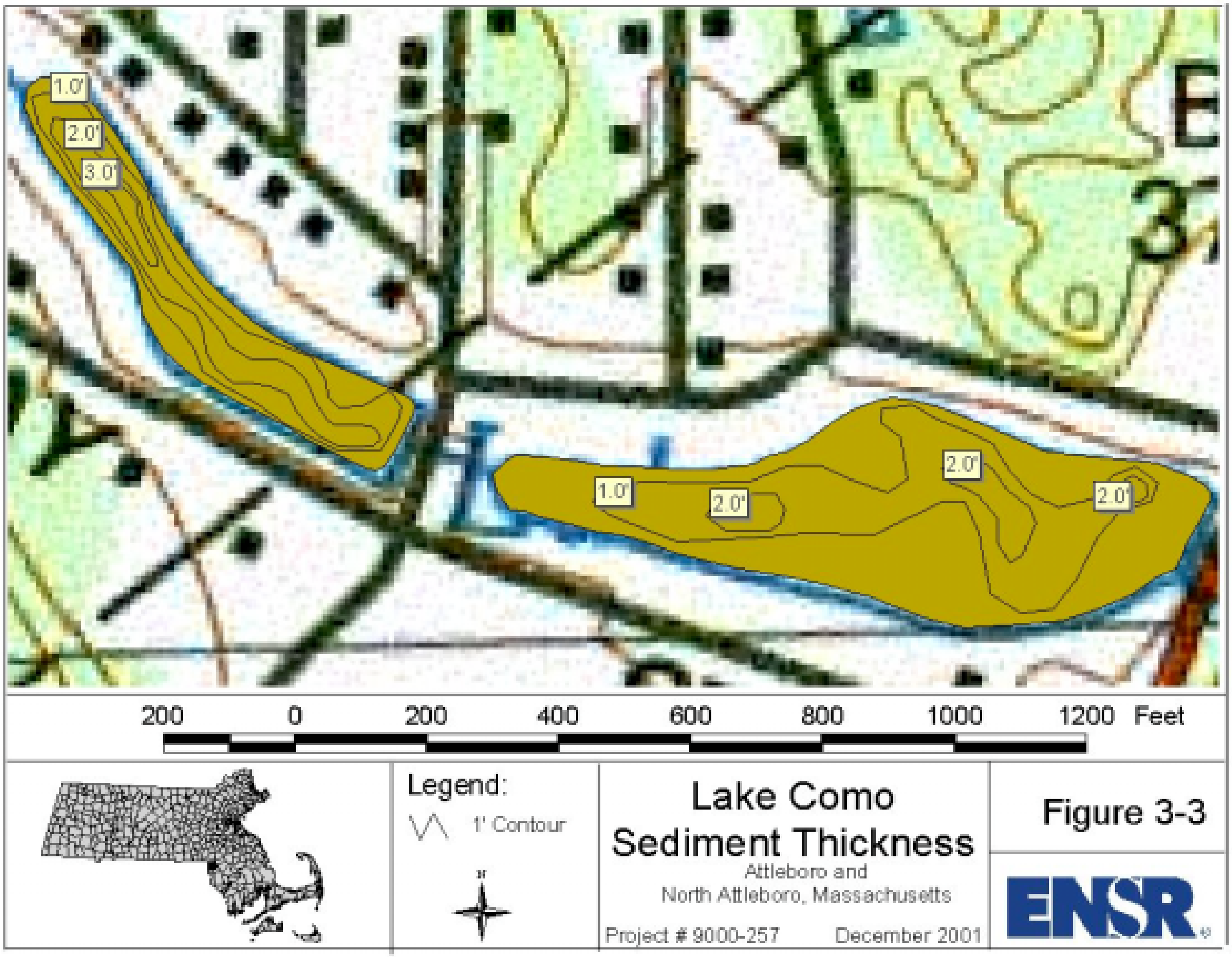
Lake Como
Bathymetry
Attleboro and
North Attleboro, Massachusetts
Project # 9000-257
December 2001
Figure 3-2
ENSR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200 0 200 400 600 800 1000 1200 Feet
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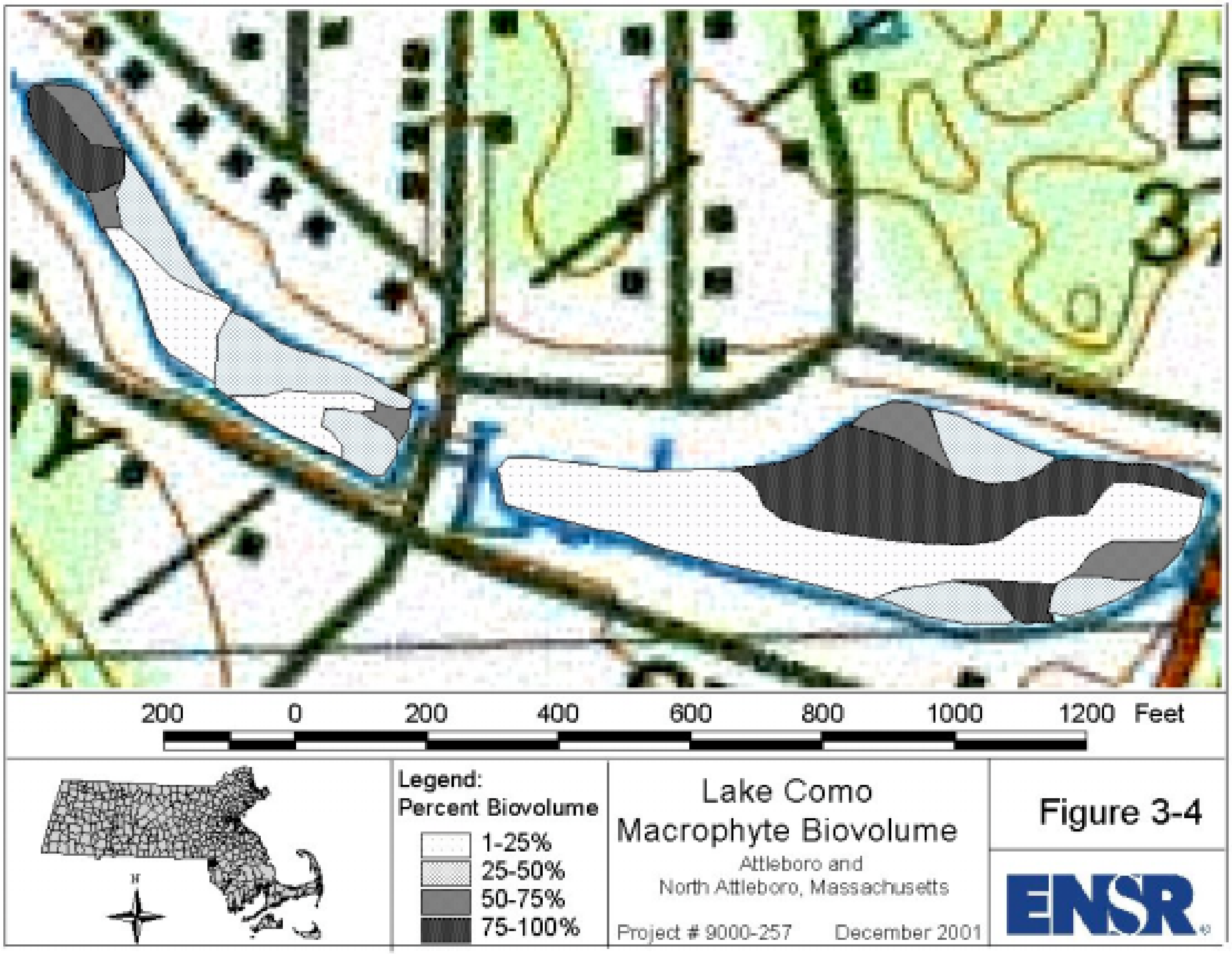
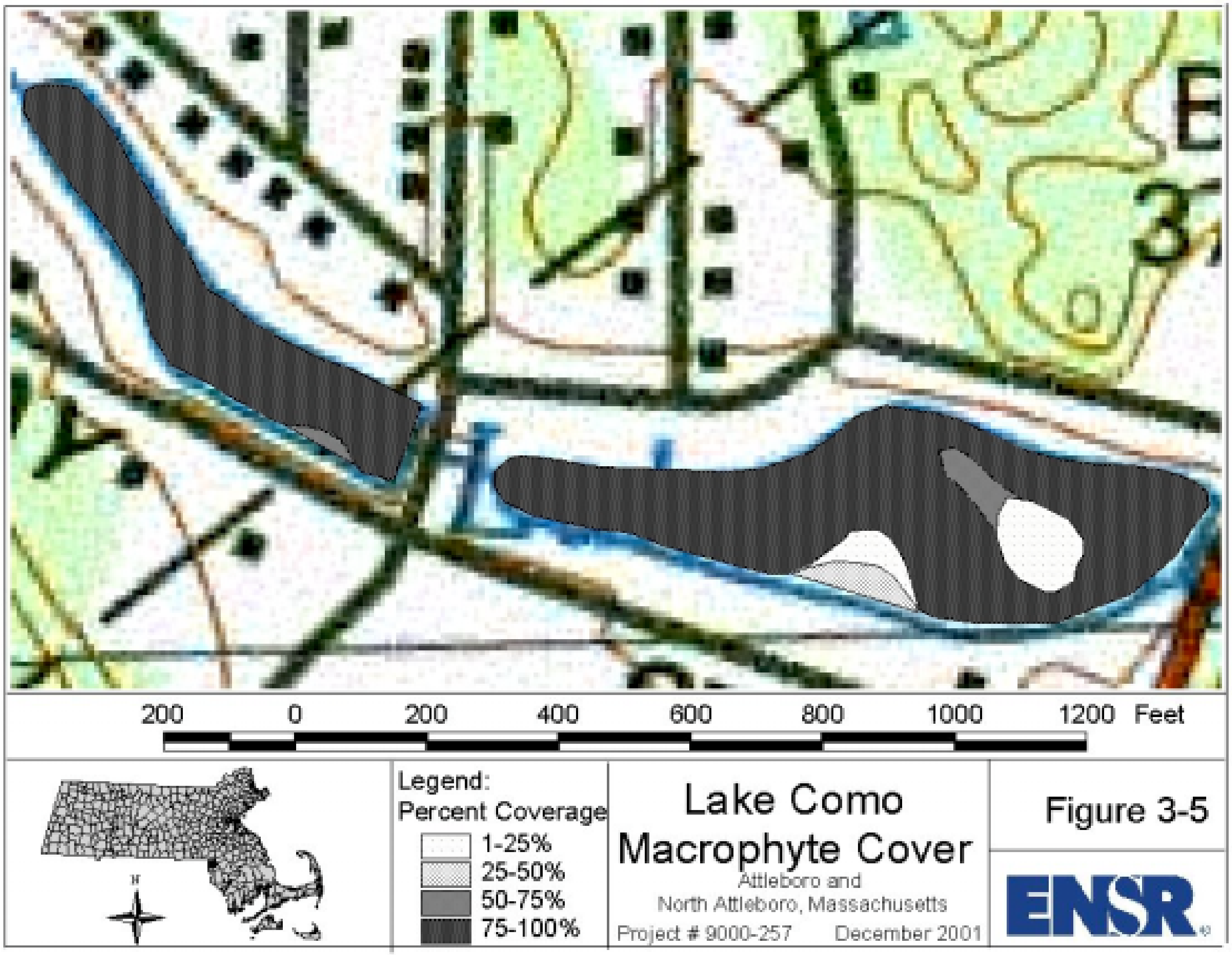
1' Contour
Lake Como
Sediment Thickness
Attleboro and
North Attleboro, Massachusetts
Project # 9000-257
December 2001
Figure 3-3
ENSR


200 0 200 400 600 800 1000 1200 Feet
Legend:
Percent Biovolume
1-25%
25-50%
50-75%
75-100%
Lake Como
Macrophyte Biovolume
Attleboro and
North Attleboro, Massachusetts
Project # 9000-257
December 2001
Figure 3-4
ENSR



200
0
200
400
600
800
1000
1200
Feet
Legend:
Percent Coverage
1-25%
25-50%
50-75%
75-100%
Lake Como
Macrophyte Cover
Attleboro and
North Attleboro, Massachusetts
Project # 9000-257
December 2001
Figure 3-5
ENSR

4.0 WATERSHED ASSESSMENT

A watershed delineation and a land use identification evaluation were conducted in support of the Lake Como restoration study. Each evaluation is described below.

4.1 Watershed Delineation and Land Use Identification

The delineation of the Lake Como watershed was completed using scanned USGS 7.5-minute topographic maps acquired from the MassGIS web site that houses geographic information for Massachusetts. The maps were imported into ArcView® and the area of the watershed was determined by tracing the topography of the terrain that contributes flow to the outlet of Lake Como. The watershed area was divided into landuse types based on the MassGIS database of landuse.

Storm drains were identified during a site inspection, and using information provided by the City of Attleboro Conservation Commission and the City of North Attleboro Conservation Commission. Because each storm drain was designed to receive overland runoff from a specific tract of land, the contributing areas were determined either by storm drain permit inspection or topographic delineation of the area upgradient of each drain. No storm drains were identified for the West Pond. Three storm drains and a small channel were identified for the Main Pond.

The area of the watershed contributing to the outlet of Lake Como is illustrated in Figure 4-1 and was estimated to be 200 acres. The watershed is approximately 5,700 feet in length and has an average width of approximately 1,500 feet. The watershed has a southeasterly aspect. Landuse in the basin is predominantly residential except for the forested uplands and the Fuller Memorial Hospital, located to the south of Lake Como.

There are four storm drain systems that discharge to the Main Pond of Lake Como: the Esker Village drain, the Fuller Memorial Hospital drain, Heather Street drain from North Attleboro, and the unnamed channel discharging from Attleboro. The West Pond was inspected and no storm drains were identified to be discharging into Lake Como upstream of Como Drive (Figure 4-1).

- The drainage area of the Esker Village subdivision that contributes flow to Lake Como is 9.6 acres. Stormwater is collected within the subdivision in storm drains and the water is transferred to the Main Pond immediately downstream of Como Drive through a concrete pipe.
- The area of the Fuller Memorial Hospital that contributes stormwater to the lake is 24 acres. Similarly stormwater is collected in storm drains and is transferred to a clay pipe that traverses May Street. This pipe is at least half filled with sediment and debris and flow through is severely restricted. Once stormwater from the hospital is on the north side of May Street it can flow overland to the Main Pond if the flow rate is sufficient.

- The area drained by the Heather Street drain in North Attleboro drains a residential area north of Lake Como and has an area of approximately 20 acres. This area is networked with a series of storm drains and storm sewer pipes to collect water throughout the residential area. Flow is ultimately transferred to Lake Como Main Pond through a small clay pipe. The pipe enters the lake immediately downstream of Como Drive.
- There is a fourth stormwater tributary to the Main Pond. The unnamed tributary is located on the north side of the main pond just west of Glen Street. This tributary appears to drain an approximately 5-acre area and is not associated with any storm sewer discharge. The location for each of the four aforementioned stormwater discharges is indicated on Figure 3-1.

4.2 Landuse and Historical Changes

In establishing landuse characteristics, ENSR evaluated MassGIS data that reports on landuse data from 1999. According to this data set, the landuse is approximately 42% low-density housing and 38% forested. Slightly less than ten percent of the land is designated open urban area (i.e. parks, easements, etc.) and slightly more than 5 percent of the watershed area is water. The remaining land is commercial, pasture, cropland, and wetland (0.68%, 0.56%, 1.3% and 1.5%, respectively).

As evidenced by the series of historical photographs (Appendix C), the uses of land in the Lake Como watershed has changed substantially over the last 40 years. This is largely attributable to new housing. The following bulleted list details some of the observations made during a comparison of the most recent aerial photo and the aerial photo dated April 22, 1959.

- On the north side of Lake Como, approximately 39 houses were counted in the 1959 photo. In the more current photo, ENSR identified approximately 64 houses in the same neighborhood. Most of these houses were built on Campus Road and Loomis Street.
- On the south side of Lake Como, along the part of May Street shown, there were approximately 7 houses in 1959. The current photo shows approximately 15 houses along the same stretch of May Street.
- The vegetation observed along the bank edges of Lake Como in 1959 appears to be groomed low grasses with a smaller percentage of wetland plants, trees, and bushes compared to the current photo.
- The hospital grounds in 1959 included one large building, several small buildings, a small parking lot and some cropland to the west of the hospital buildings. Since 1959, the hospital has expanded to approximately 8 more buildings, an additional parking lot, and a new driveway off May Street. The croplands to the west of the hospital buildings are now the new Esker Village neighborhood, at least 20 new houses.

- At the end of Campus Drive, there is a U-shaped feature that in 1959 appears to be a pond, and in the current photo appears to be a dry, shallow depression. The depression appears dry in the 1971 photo and wet again in 1981.
- In the 1959 photo, there appear to be two stream channels at the headwaters of the west pond. The northern channel is wide and relatively open, compared to the southern channel, which is very straight and appears to be man-made. The more natural northern drainage feature is not evident in the current photo. The straight channel is less distinct than in the past but appears to be the dominant path of drainage from the upper, West Pond wetlands. The progressive overgrowth of the wider channel can be observed by looking at the pictures in series. It is not known who maintained this channel in the past.
- The difference in the area above the headwaters is minimal. There appear to be no new roads leading to new construction or development of the uplands to the west and north of the ponds.

The steady addition of residential homes and expansion of the Fuller Hospital area are likely two main sources of increased rainfall runoff and resulting nutrient loading to Lake Como. The apparent redefining and straightening of the inlet channel has likely reduced the travel time and thus, the filtering time by the wetland plants of the nutrient- rich water.

4.3 Description of Public Use Areas

A lakeshore reconnaissance survey was conducted to identify public use or access areas to Lake Como. One public use area was identified in the form of a small park along the southern shore of Main Pond. The park extends approximately 500 feet along the southern shoreline across from the Hospital. The park includes a park bench, a narrow grassy area and a small semi-circular. The park does not include a boat launch, but a clearing exists allowing for small boat access (e.g., a canoe).

4.4 Watershed Assessment Summary

The Lake Como watershed assessment resulted in the following observations:

<u>Physical Setting.</u> The Lake Como drainage area is approximately 200 acres in size. The watershed is approximately 5,700 feet in length and has an average width of approximately 1,500 feet. The watershed has a southeasterly aspect. Landuse in the basin is predominantly residential except for the forested uplands and the Fuller Memorial Hospital located to the south of Lake Como.

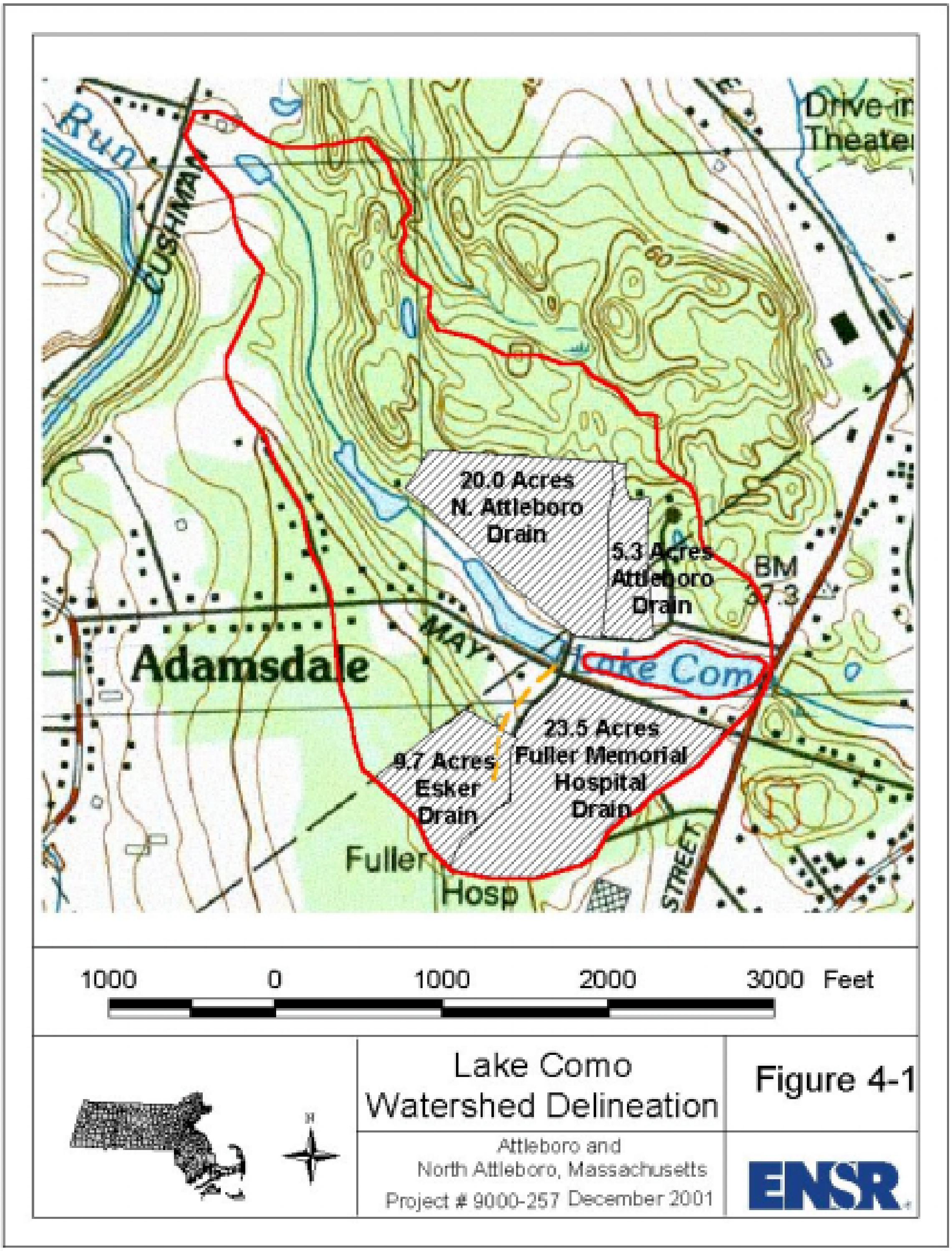
<u>Landuse.</u> Based on the comparison of aerial photos dating back approximately 40 years to recent photos and field observations, we can conclude that overall the runoff characteristics of the watershed have most likely changed as a result of the increase in number of houses in this area. In the southeast portion of the watershed, the periodic additional development at Fuller Memorial Hospital is also likely

to cause changed runoff characteristics. Typically these kinds of developments increase runoff and nutrient loads to downstream receiving waters.

Drainage. A natural channel that flows south-southeast from the inlet through wetland to the West Pond drains the watershed. A culvert conveys water from the West Pond to the Main Pond underneath Como Drive. Four drains convey storm flow from the upper portions of the watershed to the Main Pond of Lake Como. They are as follows:

- Esker Village storm drain. Drains southwest subdivision. Approximately 9.6 acres.
- Fuller Memorial Hospital (Hospital) storm drain. Drains southeast urban open area. Approximately 24 acres.
- North Attleboro storm drains. Drains a residential area northwest of Lake Como. Approximately 20 acres.
- Attleboro drain. Drains a very small approximately 5-acre area and is not associated with any storm sewer discharge.

Results of the watershed assessment were applied to support the Lake Como watershed modeling application described below.


Drive-in
Theater
Run
CUSHMAN
20.0 Acres
N. Attleboro
Drain
5.3 Acres
Attleboro
Drain
BM
37.3
MAY
Adamsdale
Lake Como
23.5 Acres
Fuller Memorial
Hospital
Drain
9.7 Acres
Esker
Drain
Fuller
Hosp
STREET
1000 0 1000 2000 3000 Feet
N
Lake Como
Watershed Delineation
Figure 4-1
Attleboro and
North Attleboro, Massachusetts
Project # 9000-257 December 2001
ENSR

5.0 SCREENING LEVEL WATERSHED MODELING APPLICATION

A screening level watershed model based on landuse was applied to simulate average annual flows and nutrient loads in the Lake Como system. The model was developed by ENSR and features application of several widely accepted watershed process estimation methods including those of Vollenweider (1968 and 1975), Kirchner and Dillon (1975), and others. A brief description of the watershed modeling application is provided below and a detailed description is provided in Appendix D.

The Lake Como watershed model applies landuse-specific parameter values, such as runoff and export coefficients (Reckhow et al., 1980), to estimate the load of water and nutrients to Lake Como. In-lake nutrient concentrations may then be estimated and the extent of nutrient reductions necessary to achieve water quality improvement may be evaluated. Application of the screening level watershed model consists of a series of steps designed to allow the user to replicate the current watershed and in-lake situation and then evaluate alternative management scenarios to improve in-lake water quality. The following watershed modeling steps were undertaken as part of the Lake Como watershed evaluation.

1. Delineation of watershed and subbasin boundaries

2. Determination of average flow using runoff coefficients

3. Determination of nutrient loads using export coefficients

4. Estimation of in-lake nutrient concentrations

5. Estimation of nutrient goals in terms of permissible and critical nutrient concentrations

Each of these modeling steps is described in Appendix D including associated governing equations and literature sources.

5.1 Model Setup

Setup of the landuse based watershed model requires knowledge of the individual subbasins within the overall drainage basin, including the size of these watersheds, landuses within them, the overall drainage pattern, and any other features that could increase or decrease nutrient loads. The Lake Como watershed was divided into subbasins that each represent the inflow from tributaries, storm drains and direct groundwater flow to Lake Como. The landuses for each of the subbasins was determined using information from the Massachusetts GIS Internet site (http://www.magnet.state.ma.us/mgis/), with field inspection for verification. Precipitation data was obtained from the National Weather Service gauging station in Boston, MA and applied to the Lake Como watershed modeling application.

Median precipitation runoff or export coefficient values for each land use in the Lake Como watershed were initially selected and applied based on available data and best professional judgment. Attenuation coefficients were selected according to expected removal efficiency associated with mapped features of the watershed and stream system. Information on lake features was derived from maps and field surveys.

5.2 Model Development

The model development process involved applying the screening-level Lake Como watershed model, and adjusting model parameter values, to improve the accuracy of flow and nutrient load predictions of current conditions. Water quality data collected during the field investigation were applied to develop and establish a crude calibration of the screening level watershed model. Nutrient concentrations predicted by the model at the discharge point in each watershed were compared to available field data. Model parameter values were also compared with appropriate literature values to insure that reasonable values were applied to the model. The model development process resulted in selection of runoff, export, and attenuation coefficient values, that collectively resulted in reasonable representations of current Lake Como watershed processes on an average annual basis. Development of the screening level landuse model was designed to achieve the following two goals:

1) Adjust watershed export and attenuation coefficients within literature-defined range for each landuse or attenuator type so an optimal match between model predicted and field measured nutrient concentrations, at key points in the system, is obtained.

2) Establish an optimal match between model predicted and field measured in-pond nutrient concentrations.

Model development was successfully completed. Model predicted and actual in-stream values were acceptably similar. It should be noted that this is a screening level model and model predictions should be interpreted as rough estimates of actual site conditions. The predictions are very useful, however, for evaluating and comparing expected water quality improvements associated with various pond restoration alternatives.

The model was applied to predict flows and nutrient concentrations associated with each stormwater source and tributary. Annual average flows and nutrient loads were then predicted using the precipitation record for the area obtained from the National Weather Service gauge in Boston, MA.

5.3 Hydrologic Simulation Results

The Lake Como model was applied to estimate mean turnover rate and residence time and to estimate the relative contribution of each water source to the Lake Como system. Each of these hydrologic calculations is described below.

5.3.1 Lake Como Mean Turnover Rate and Residence Time

Mean turnover rate and residence time values are hydrologic metrics describing the relationship between the volumetric rate of water entering the pond and the total pond volume. Turnover rate and residence time values can provide insights on the quality of a pond habitat and its ability to support diverse animal and aquatic plant communities.

Turnover rate is defined as the number of times that the total volume of water in the lake is exchanged during a unit period of time. The turnover rate is defined as the volumetric rate of water entering the lake divided by the lake volume. Water entering the Lake Como was represented as watershed runoff and neglected groundwater inflow/outflow and evaporation processes. Turnover rates may be estimated on an annual and/or seasonal basis since the effects of the turnover rate are usually the result of seasonal differences in precipitation. Shorter-term calculations can be made to identify the turnover rate associated with a particular storm. Residence time is defined as the inverse of turnover rate and expresses the period of time that a water parcel remains in the pond assuming all water moves through the pond at the same rate.

Table 5-1 presents turnover rate and residence time estimates in Lake Como for each of four periods including the spring, summer, annual, and for a 24-hour storm with a 1-year return period. Runoff rates were determined by multiplying the average annual runoff coefficient by the runoff duration. The average annual runoff coefficient was determined by dividing the average annual runoff (33.5 in/yr), determined in the spreadsheet model, by the average annual precipitation (44.9 in/yr).

During the spring and summer seasons, Lake Como turns over approximately 5 times with a mean residence time of 17 days. On an annual basis, the lake is estimated to turn over 24 times with a mean residence time of 15 days. Lastly, a major precipitation event, the one-year storm, results in one complete exchange of water in the Lake Como system – flushing out the entire lake volume in one day.

5.3.2 Flow Budget for Lake Como Watershed

The Lake Como screening level watershed model was applied to estimate the flow budget of Lake Como on an average annual basis. The flow budget was developed using the results of the watershed model under present conditions. As described in the watershed assessment (Section 5), the Lake Como watershed may be represented as six separate subbasins for the purposes of watershed landuse-based modeling. These subbasins included the areas contributing to the North Attleboro, Attleboro, Hospital, and Esker Village stormdrains, and the remaining West and Main Ponds.

The estimated annual flow budget for the Lake Como watershed is summarized in Table 6-2. The flow budget developed using the Lake Como spreadsheet model indicates that approximately 72% of the 689 acre-ft/yr of precipitation (44.9 in/yr) was accounted for as direct runoff. Additionally, the annual ET accounts for 30% of the water loss from the basin. The groundwater inflow to the basin was accounted for by calculating the difference between the precipitation and the runoff and ET from the basin. Groundwater inflow amounted to a loss of 2% from the water budget.

5.4 Water Quality Simulation Results

The Lake Como model was applied to estimate the relative contribution of nutrient loads for each water source to the Lake Como system. The nutrient load estimation process was conducted on an annual average basis for current conditions. Next, phosphorus reduction goals were established for the Lake Como system based on a widely accepted assessment method. The screening-level Lake Como watershed model was then applied to evaluate the potential for each of several restoration management alternatives to improve water quality in the Lake Como system. Each of these water quality assessment tasks is described below.

5.4.1 Assessment of Present Phosphorus Loads

The Lake Como screening level watershed model was applied to estimate the phosphorus budget of Lake Como on an average annual basis. The phosphorus budget was developed using the results of the watershed model under current conditions. Phosphorus loads were derived for each of the six subbasins within the Lake Como watershed. External sources of phosphorus were assumed to include atmospheric loading and a load from waterfowl. Phosphorus loads from each subbasin component were predicted by applying landuse based export coefficients to each of the subbasins. Phosphorus concentrations were then calculated by dividing the predicted phosphorus load by the predicted volumetric flow rate from each subbasin (presented in Section 5.3.2 above). The atmospheric load of phosphorus to the surface of Lake Como was assumed to be 0.4 kg/yr and the waterfowl load to the lake was estimated to be 3 kg/year.

The estimated annual flow budget for the Lake Como watershed is summarized in Table 5-3. The predicted phosphorus load from each source indicates that nearly all sources of phosphorus to Lake Como come from the surrounding subbasins (96%). Subbasin areas contributing the most annual phosphorus load were the areas contributing to the West Pond (31%), the Hospital stormdrain (21%), and the North Attleboro stormdrain (19%). Phosphorus contributions from the remaining three subwatersheds were considerably less (5-12% each) and loads from external sources were comparatively minor (<4%).

5.4.2 Phosphorus Reduction Goals for the Lake Como System

An assessment method developed by Vollenweider (1968) was applied to Lake Como to provide an estimate of the extent of nutrient reduction that would likely be necessary to remove water quality and biological impairment from the Lake Como system. Permissible and critical limits for phosphorus loads were estimated where the permissible load is defined as the amount of phosphorus that could enter a system without having obvious or continual detrimental effects. In other words, if permissible levels of nutrients are present, then the waterbody is not likely to be impaired. The critical limit is defined as twice the permissible load. If critical levels of nutrients are present, then the waterbody is likely to be impaired. In general, as nutrient concentrations exceed the permissible load level and approach the critical load level, nuisance algal blooms and other aquatic biological activity can become problematic. Lakes exceeding the critical load typically experience serious productivity problems.

For the Lake Como application, the Vollenweider method was applied and permissible and critical phosphorus loads for Lake Como were calculated to be 11 and 22 kg/yr, with corresponding in-lake concentrations of approximately 16 and 31 µg/L, respectively. The permissible and critical nutrient load estimation process is a screening level tool and additional factors, such as the physical characteristics of the lake system should also be evaluated.

A review of phosphorus measurements collected during the Lake Como field investigation indicates an average in-lake phosphorus concentrations averaged 90 µg/L (see Table 3-1 summary statistics). The Lake Como watershed model predicted total annual phosphorus loadings of 109 lbs/yr. Phosphorus level in the ponds and phosphorus loads from the watershed are both well above the predicted permissible and critical phosphorus concentrations and suggests that phosphorus levels must be reduced by approximately a factor of 5 before water quality impairment may be removed from Lake Como. Thus, restoration alternatives will need to be selected and applied to reduce phosphorus concentrations dramatically, if water quality standards and designated use goals are to be attained.

5.4.3 Model Simulation Results for Restoration Alternatives

Presently, phosphorus concentrations in Lake Como are sufficient to lead to water quality impairment. The Lake Como watershed model was applied to evaluate several restoration alternatives for reducing nutrient loads. The watershed model is designed to evaluate watershed runoff of water and nutrients and to estimate in-lake nutrient concentrations, given specified physical characteristics. Thus, the model is capable of estimating water quality improvements associated with modifications to sources of nutrients. The model is not well suited however, to provide accurate predictions of water quality effects associated with other restoration alternatives, such as modifications to impoundment dams or dredging. These restoration alternatives are important for Lake Como and will be discussed in "Section 6 - Management Goals and Recommendations", but are not evaluated using the watershed model.

A total of 4 restoration alternatives were simulated using the watershed model and predicted reductions in total nutrient loading were obtained. Predicted reductions in phosphorus loads are relative to present loads (presented in Table 5-3). Descriptions and modeling results for each restoration alternative scenario are provided below.

Scenario 1: Remove All Storm Drain Flow. This scenario specifies redirecting all storm drain flow out of the watershed (downstream of Main Pond watershed). This restoration alternative would require rerouting storm drain flow via a set of pipes to connect stormdrains to the Route 1 storm drain network, just below the main pond dam. An engineering feasibility assessment would likely be required to evaluate a set of options for removing all storm drain flow and to estimate the total cost of the project.

The scenario 1 restoration alternative was predicted to result in a 50% decrease in total annual phosphorus load to Lake Como.

Scenario 2: Infiltrate All Storm Drain Flow. This scenario specifies installation of an infiltration system for all storm drains to attenuate nutrient load to the waterbody. This restoration alternative would require construction of detention basins and/or dry wells to enable stormwater to infiltration to groundwater, rather than being transported directly to Lake Como. Stormwater infiltration systems have been demonstrated to be highly effecting in removing nutrients from storm flow.

The scenario 2 restoration alternative was predicted to result in a 40% reduction in total annual phosphorus loading to the Lake Como system.

Scenario 3: West Pond Wetland Conversion. This scenario specifies that the West Pond be allowed to revert to a wetland. This restoration alternative would increase the capacity of West Pond to attenuate nutrients prior to being released into the Main Pond. The tradeoff associated with this alternative involves allowing West Pond to degrade from a pond to a wetland in exchange for improving water quality in the Main Pond.

The scenario 3 restoration alternative was predicted to result in a 10% reduction in total annual phosphorus loading to Lake Como. Other processes not accurately simulated by the model, such as retention of nutrients by wetland vegetation, and hydrologic modification in West Pond, may result in greater phosphorus load reductions than those predicted.

Scenario 4: West Pond Wetland Conversion and Infiltrate Storm Drain Flow. This restoration alternative is a combination of Scenarios 2 and 3 above – featuring conversion of West Pond to a wetland and installation infiltration system for all stormwater drainage.

The scenario 4 restoration alternative was predicted to result in a 50% reduction in total annual phosphorus loading to Lake Como.

Summary of Model Scenarios

The model was applied to evaluate restoration alternatives for stormwater drains and tributaries in the Lake Como system. Restoration management alternatives featuring removal of stormdrain discharges or groundwater infiltration of discharges were predicted to result in 40% to 50% reductions in total annual phosphorus loads to Lake Como. These reductions alone would not be sufficient to substantially reduce water quality impairment, based on phosphorus reduction goals for the Lake Como system. A combination of restoration alternatives featuring watershed nutrient reductions, as described above, and other alternatives, such as dam repair and dredging, may be required to achieve the necessary water quality improvements in the Lake Como system. A discussion of management alternatives and recommendations is provided below.

5.5 Summary of Watershed Modeling Results

The watershed modeling application provided the following insights on the Lake Como system:

- Pond Hydrologic Assessment - The hydrologic assessment of Lake Como indicated that the mean turnover rate in Lake Como is approximately 5 times per season during the spring and summer (residence time 17 days). Also, the one-year storm was estimated to result in one complete replacement of pond volume (residence time 1 day).

- Watershed Phosphorus Load Budget – The average annual phosphorus load was estimated to be approximately 105 kg/yr. The sources of phosphorus were as follows (with % contribution shown: 31% from runoff directly to West Pond (via tributary and overland); 21% from the Fuller Hospital storm drain; 19% from the North Attleboro storm drain; 12% from runoff directly to the Main Pond; 8% from the Esker Village storm drain; and 5% from the Attleboro storm drain.

- Target Phosphorus Loading and In-Lake Goals - Widely accepted methods were applied to develop target phosphorus load and in-lake goals. A total annual phosphorus load goal of 11 to 22 kg/yr was established and an in-lake phosphorus concentration goal of 16 to 31 μg/L was established. Comparison of phosphorus targets to watershed model predictions of phosphorus loads and field measurements of in-lake concentrations indicates that phosphorus concentration must be reduced by approximately a factor of 5 (i.e., 5 times less load) before water quality improvement may be achieved.

- Evaluation of Restoration Alternatives – Restoration alternatives featuring reduction or removal of storm drain loads were evaluated using the watershed model. Removal or infiltration of storm drain flows from the system was predicted to result in 40% to 50% reductions in average annual loads of phosphorus to the ponds. These modifications would have to be implemented along with additional modifications in order to achieve necessary water quality improvements.

The management goals and recommendations section, below, provides a complete description of recommended restoration activities for the Lake Como system.

Table 5-1   Lake Como Hydrology: Estimated Turnover Rates and Residence Times

Duration	Total Precipitation (in)	Residence Time (days)	Number of Turnovers
Spring (3 months)	10.2	17	5.5
Summer (3 months)	9.8	17	5.3
Annual	44.9	15	24.3
24-hour 1-year storm	1.7	1	0.9

Table 5-2 Lake Como Hydrology: Estimated Water Budget

Parameter	Value	Percent
Precipitation (acre-ft/yr)	689	100%
Total Runoff (acre-ft/yr)	497	72%
• North Attleboro Storm Drain (acre-ft/yr)	• 67	• 10%
• Attleboro Storm Drain (acre-ft/yr)	• 17	• 3%
• Hospital Storm Drain	• 59	• 9%
• Esker Village Storm Drain	• 33	• 5%
• Contribution to West Pond	• 268	• 39%
• Contribution to Main Pond	• 52	• 8%
[1]Evapotranspiration (acre-ft/yr)	207	30%
[2]Groundwater (acre-ft/yr)	(15)	(2%)
[1]Half of Potential ET (from Water Atlas, 1973) [2]Precipitation = Runoff + ET + Groundwater		

Table 5-3 Lake Como Water Quality: Estimated Phosphorus Budget

Phosphorus Loading Component	Value	Percent
Atmospheric Deposition (kg/yr)	1.2	1.1%
Total Watershed Load (kg/yr)	105	96%
• North Attleboro Storm Drain (kg/yr)	• 21	• 19%
• Attleboro Storm Drain (kg/yr)	• 5	• 5%
• Hospital Storm Drain (kg/yr)	• 23	• 21%
• Esker Village Storm Drain (kg/yr)	• 9	• 8%
• Contribution to West Pond (kg/yr)	• 34	• 31%
• Contribution to Main Pond (kg/yr)	• 13	• 12%
Waterfowl (kg/yr)	3	2.8%
Total Load of Phosphorus to Lake (kg/yr)	109	100%

6.0 MANAGEMENT GOALS AND RECOMMENDATIONS

Observations of existing problems, documented as part of the Lake Como Restoration Study, are stated below followed by a statement of lake restoration goals. The description of existing problems and goals provides context for presentation and discussion of a set of management recommendations for the Lake Como system presented in Section 7.3 below.

6.1 Observations of Existing Problems

The following existing problems were observed in the Lake Como system:

- Low water levels during the summer in the main basin.
- Excessive concentrations of phosphorus, nitrogen, and other pollutants in stormwater entering the ponds.
- Excessive concentration of phosphorus, nitrogen, suspended solids and other pollutants in the water column in the ponds.
- Extensive aquatic vegetation primarily aquatic macrophytes on the water surface and in the water column of the ponds
- Soft, nutrient-rich sediments supporting growth of rooted plants that impede recreational activities.

6.2 Lake Como Restoration Goals

Lake Como restoration goals include the following:

- Improve water quality to enable the ponds to achieve the water quality standards associated with its designated use. Specifically, enable the ponds to meet numeric criteria for ambient dissolved oxygen concentration and narrative criteria for nuisance aquatic vegetation;
- Enable passive uses of the ponds for non-motorized boating;
- Improved visual aesthetics, with reduced odor and unsightly growths; and
- Restore the habitat of the ponds to support fish and wildlife populations.

Restoration management plans designed to support achievement of these goals are described below.

6.3 Management Recommendations

A complete management program should include watershed and in-lake components, as individual components do not appear to provide for optimal long-term conditions in Lake Como. ENSR has evaluated a variety of management options for Lake Como and identified the following 4 recommended restoration tasks:

1. Repair leaky dam to support increased pond volume and water level.

2. Dredge nutrient-rich pond sediments to reduce sources of excess biological growth and increase pond volume.

3. Reduce nutrient loading to ponds through infiltration or removal of stormwater sources.

4. Develop a watershed education program to provide interested parties with information necessary to reduce point and non-point sources of nutrients in the Lake Come watershed.

A description of each of these 4 recommended restoration alternatives is provided below. All alternatives explored as part of the Lake Como Management Alternatives Analysis are provided in Appendix E of this report. Included in Appendix E are also some suggested Best Management Practices that could be employed within the watershed (i.e. catch basin clean outs, public education, etc.).

6.3.1 Repair of Leaky Dam to Maintain an Appropriate Pond Water Level

Repairing the outlet structure is a logical option for raising the water level and improving habitat in Main Pond; however, ownership of the dam will need to be determined by the City of Attleboro. Reconstruction of the outlet along Route 1 would facilitate up to a 3 foot increase in the water level in a cost-effective manner. Repair of the outlet structure should include construction of an antechamber with flashboards in front of the current outlet area. This design would provide flexibility to manage the pond in response to flood conditions and to keep water levels as high as possible during dry periods. The repaired outlet structure would 1) normally hold the water level close to full level, 2) minimize water level rise during storms, and 3) facilitate drawdown for maintenance or plant control by flashboard removal.

Installation of such a new outlet would constitute a major overhaul of the existing structure, but could be permitted as dam maintenance as it would occupy almost the same footprint and have the same intent as the original structure. Permitting through the Attleboro Conservation Commission under the Wetlands Protection Act is essential, and Section 401 and 404 permits may or may not be necessary, depending upon regulatory interpretation. No Great Pond provisions of MA law apply, and it seems unlikely that the current dam is even listed as a permitted structure. Filing through the Dam Safety Unit of DEM may not be necessary, but that group should be contacted for input. The cost of such an upgrade is difficult to estimate at this time, but might be expected to be in the range of $50,000 to $100,000, inclusive of design and permitting expenses.

6.3.2 Dredge to Reduce Excess Growth of Aquatic Vegetation

While raising the water level will be a definite aid to improving pond appearance, the existing organic sediment deposits and nutrient-rich influent will continue to support dense plant growths and continued algal mat production. Dredging the ponds, or at least the Main Pond, to remove the accumulated organic sediment is the preferred approach to restoring the ponds. Accumulated organic sediment of approximately 7,850 cubic yards in the Main Pond and 4,830 cubic yards in the West Pond should be completely removed in order to reduce plant and algal mat growths to aesthetically appealing levels. Such dredging would also improve the habitat for most desirable forms of aquatic life. Plants would not be eliminated, but densities should be reduced by approximately 75%, likely creating better open water habitat for fish and enhancing habitat value for birds, reptiles and amphibians.

Assuming that the outlet is to be reconstructed, it would make sense to drain the pond, at which time dredging could be conducted. If it is desired that both ponds be reclaimed, both could be dredged at once, but it would be logistically and ecologically appropriate to do them sequentially to provide a refuge for fish and wildlife during the process. The West Pond would be dredged first, as the upstream impoundment, followed by the Main Pond. Wet weather may limit in-situ dewatering, but it does appear that this project can be carried out without an overly sophisticated dredging plant. Two access ramps can be established for each pond and restored later to avoid any permanent damage to bank resources. Conventional excavation equipment should be able to enter each pond and work from the edge, possibly from mats at first, clearing the bottom of soft sediment. Temporary storage may be needed, and the feasibility of using the parkland or part of the adjacent hospital grounds should be explored. Sediment appears to be of sufficiently high quality to be trucked to a variety of productive use sites without regulatory problems.

In dry condition, the sediment that now occupies almost 12,000 cyds will not likely have a volume more than 3,000 to 4,000 cyds, but getting it into that dry condition will require either spreading it thinly or applying enhanced dewatering techniques. Having sufficient temporary storage or a permanent disposal site where water content does not matter (e.g., a former gravel pit) will be important to material utility and cost control.

The cost of dredging depends on a wide range of site-specific factors, but experience has shown that a cost range from $5 to $30 per cyd is typical, with higher or lower values possible where the material has substantial resale value or where contamination and distance to the disposal site are major factors. A typical rough estimate of $10 per cyd is often assumed to provide some indication of possible level of expense, suggesting a cost of about $120,000 in this case. Permitting would include an Order of Conditions under the Wetlands Protection Act and Section 401/404 permits, at a minimum.

An ENF (Environmental Notification Form) would be required for the proposed dredging project since dredged material is estimated to exceed 10,000 cyds. The Secretary would then determine if an EIR (Environmental Impact Report) would be required. However, a Request for Determination should be filed with MEPA (Massachusetts Environmental Policy Act) once the details of the project have been determined. MEPA could rule that dredging of Lake Como is "Routine Maintenance" in which case an ENF and an EIR would not be required. Therefore, selection of temporary and permanent disposal sites would be the next logical step in pursuing this option.

6.3.3 Reduce Nutrient Loads to Ponds

Reductions of nutrient loads to ponds were evaluated using the screening level Lake Como watershed model and are summarized in Section 5. Nutrient loads must be reduced, along with repair of the outlet structure and dredging, to ensure that the ponds do not quickly revert to present conditions. Failure to address incoming water quality concerns could result in only temporary improvement or in substitution of one problem for another (e.g., algal blooms instead of plant nuisances). The watershed modeling management scenarios clearly indicate that small efforts will not provide the level of improvement in water quality necessary to make a difference in the lake, at least in terms of nutrient levels. It will be necessary to manage most runoff to a high degree, although this problem can certainly be attacked over time, discharge by discharge.

Infiltration, detention, and/or redirection of storm water would appear to have the greatest potential for achieving the goal of dramatic reduction in nutrient loads to the ponds. The design of infiltration systems will depend upon the site specific conditions, most especially soil permeability and depth to groundwater. It appears that some form of lateral system may be needed for each of the four major drainage systems entering the Main Pond, to ensure infiltration of as much runoff as possible. Positioning under existing lawn areas may be appropriate, and simple leaching catch basins could be installed in some cases. Assuming that all four current discharges are addressed with leaching systems that incorporate capture of the first half inch of runoff, pre-settling basins (to minimize clogging), and a laterally oriented infiltration system to avoid any shallow ground water problems), the cost is likely to be on the order of $120,000. Simpler systems may be applicable, if soil conditions are suitable.The design of an upstream detention system must maximize detention and associated pollutant retention processes while minimizing flood problems around the detention area. Establishing a detention basin at the upstream end of the West Pond appears feasible, although permitting under the Wetlands Protection Act is not guaranteed. Detention further upstream, possibly by expanding wetlands along the stream corridor, may have greater merit. Given the watershed area upstream of the West Pond, the detention area should be about 3 acres in size. The cost of an appropriate detention system is estimated at approximately $50,000, if done as a habitat-enhancing wetland. Any land purchase costs would be in addition to the $50,000.Watershed Education Program

A watershed education program should be developed and made available to interested parties, particularly residents within the Lake Como watershed. The program should, at a minimum, offer clear and concise recommendations to area residents and businesses that will enable them to effectively reduce nutrient loads around the watershed. The program should clearly indicate the current state of the watershed and the benefits that can be achieved through the use of Best Management Practices (BMPs).

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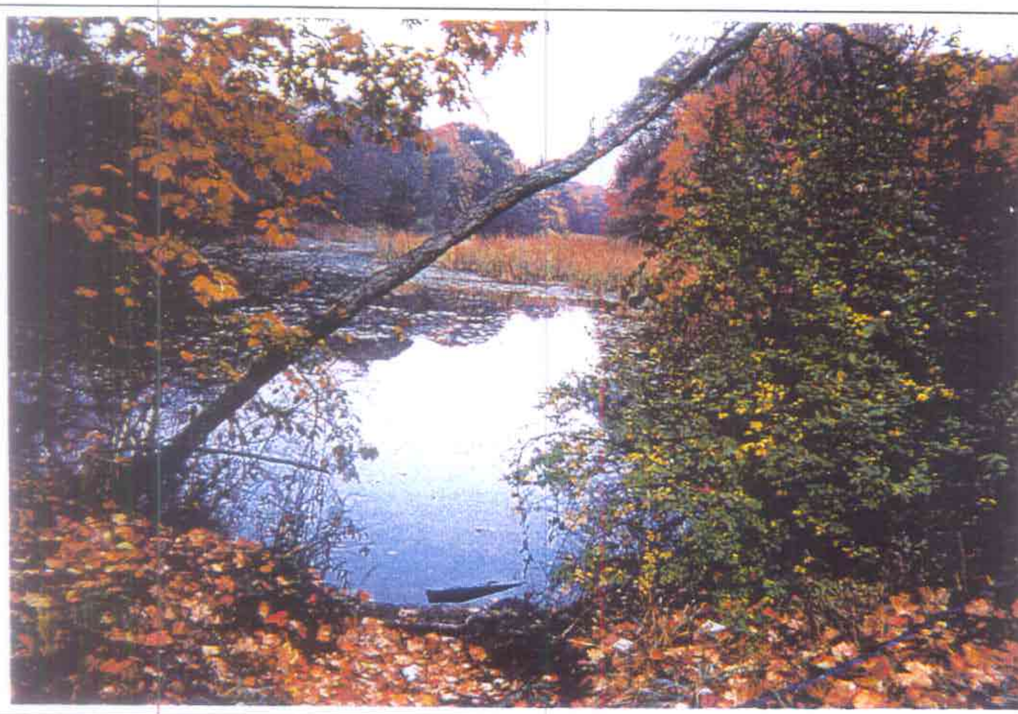
APPENDIX A

PHOTOS OF STORM DRAINS AND OUTLET STRUCTURE

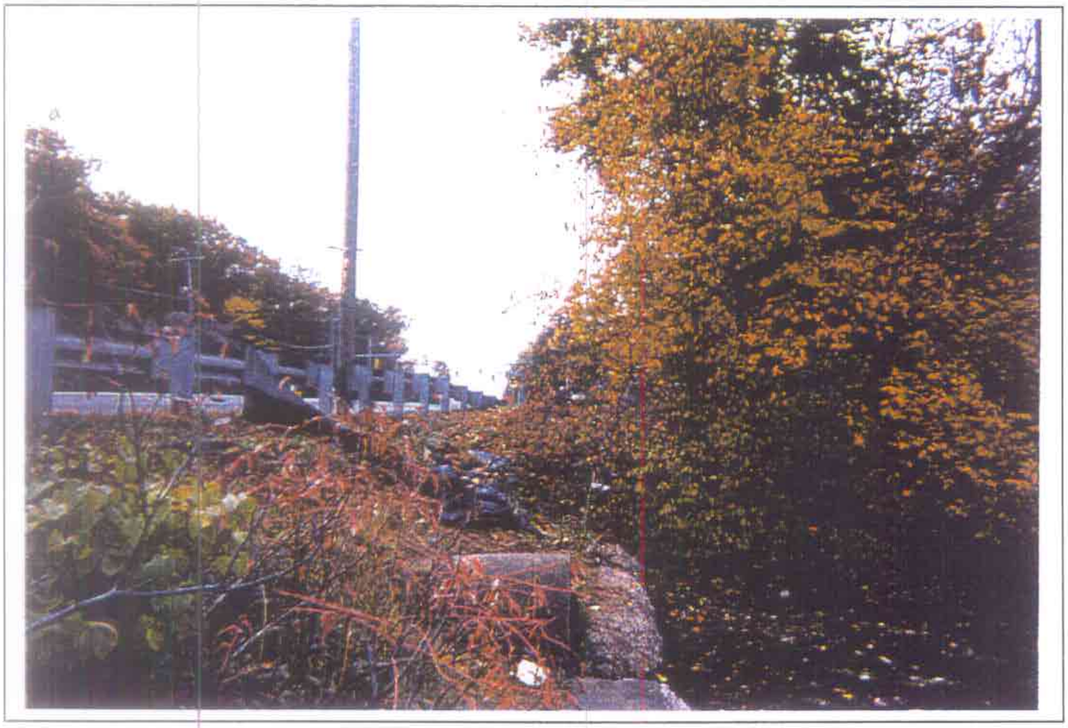


Looking West at the West Pond

Outlet of West Pond

Outlet of West Pond to Main Pond

Outlet of Main Pond (looking South)



Dam and outlet structure to Lake Como (looking South)

Outlet of Fuller Memorial Hospital Storm Drain. (looking upstream)

Inlet of Attleboro Storm Drain (looking downstream)

Inlet of North Attleboro Storm Drain

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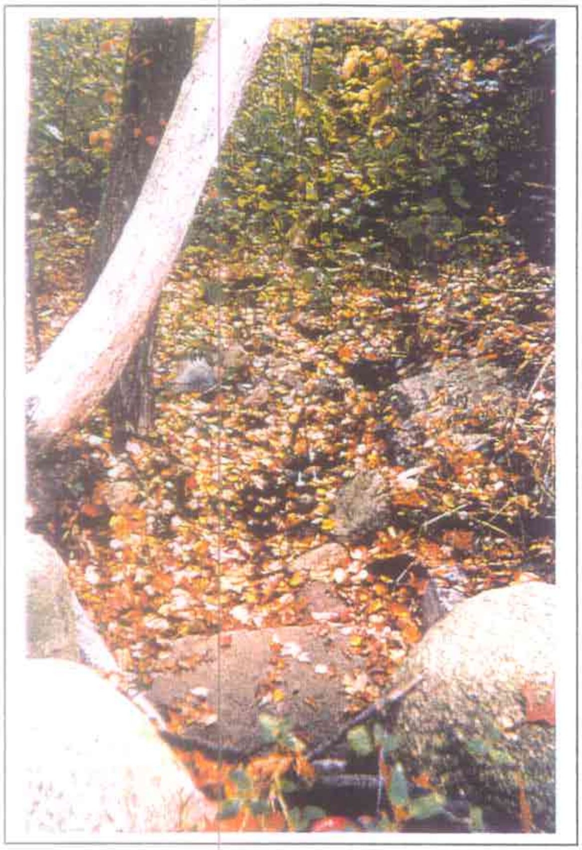

Esker Village Storm Drain (looking downstream from top of culvert)

Downstream (East) of Lake Como and Route 1

APPENDIX B

INITIAL WATERSHED-BASED POND ASSESSMENT CHECKLIST

CHECKLIST OF LAKE MANAGEMENT STUDY ELEMENTS

Lake Management study elements are listed below.

Lake Management Objectives
Existing and desired uses
Priority of uses
Special features or considerations

Watershed Features
Physical
Area, topography, drainage basins
Soils and geology
Land use and vegetation
Stormwater drainage systems
Waste disposal practices
Other pollution sources
Historical information
Flow estimates (precipitation, evaporation, surface flows, groundwater, discharges, withdrawals)

Chemical
Nutrient levels – total P, dissolved P, nitrate N, ammonium N, total Kjeldahl N
pH/alkalinity
Turbidity/suspended solids
Conductivity/dissolved solids
Temperature/dissolved oxygen
Potentially toxic compounds - anything suspected?

Lake Features
Area, shape, morphometry (depth map)
Volume, mean/maximum depths
Detention time, response time
Water chemistry (parameters as for watershed, plus water clarity)
Temperature/oxygen (top and bottom, 1 m intervals if deeper than about 15 ft)
Soft sediment distribution, physical features (muck, sand, gravel)
Sediment chemistry – N, P, selected metals, TPH, PAHs, organic content, solids content, grain size
Biology (bacteria, algae, macrophytes, zooplankton, benthic invertebrates, fish, amphibians, reptiles, birds, mammals)

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CHECKLIST OF LAKE MANAGEMENT STUDY ELEMENTS (continued)

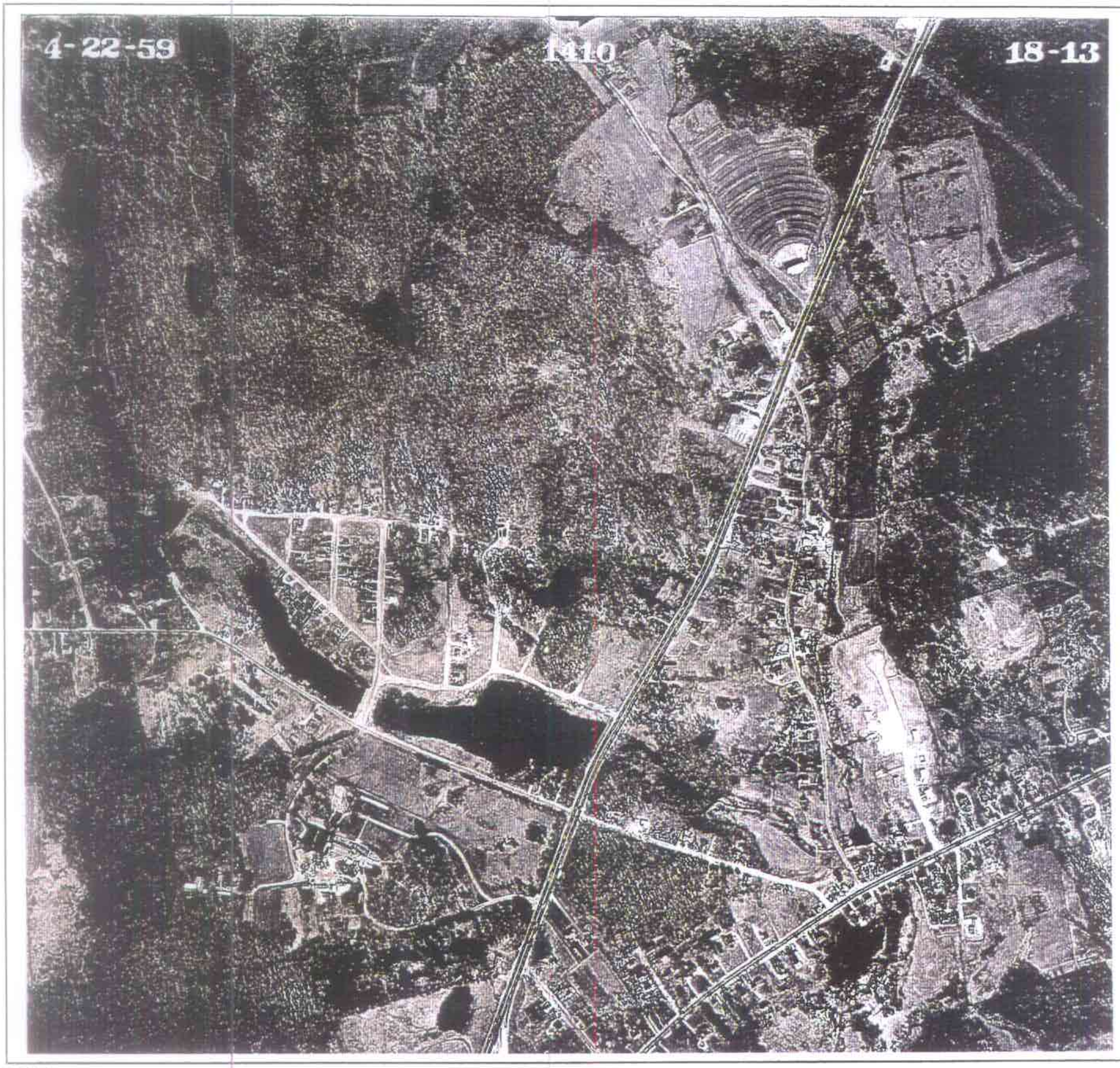
Loading Analysis (water and contaminants)
Atmospheric inputs
Dry weather surface water
Storm water
Ground water
Internal recycling (release from sediment0
Waterfowl and other wildlife
Point source discharges
Permissible/critical loads
Relationship between loads, concentrations, and algae/water clarity
Desired reduction and expected results

Management Assessment
Problem definition
Review of applicable techniques
Identification of appropriate techniques
Selection of management plan elements

Management Plan Development
Technical implementation strategy
Regulatory requirements
Cost estimates
Implementation schedule
Further data needs and monitoring program

APPENDIX C

AERIAL PHOTOS OF THE LAKE COMO WATERSHED



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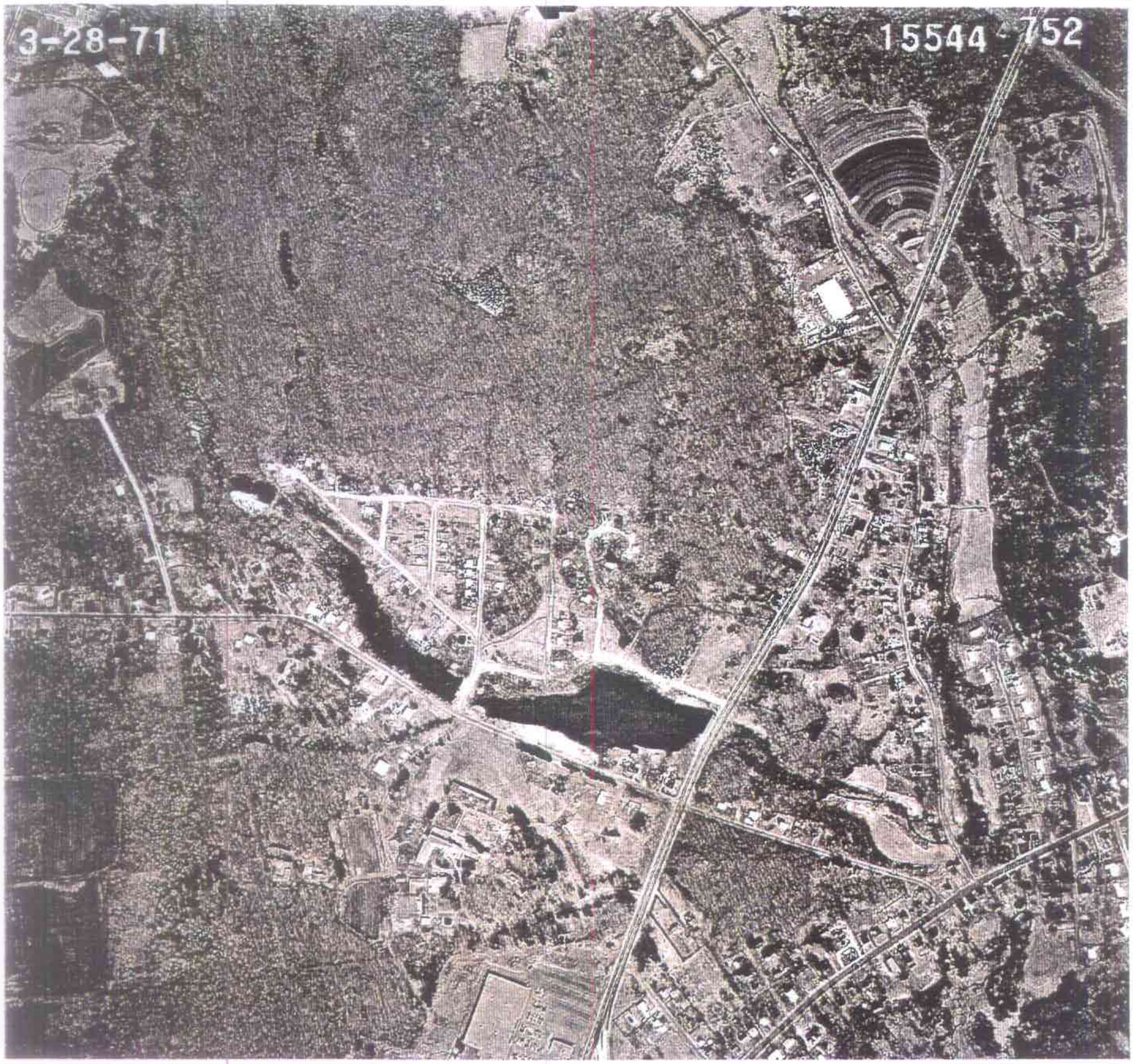

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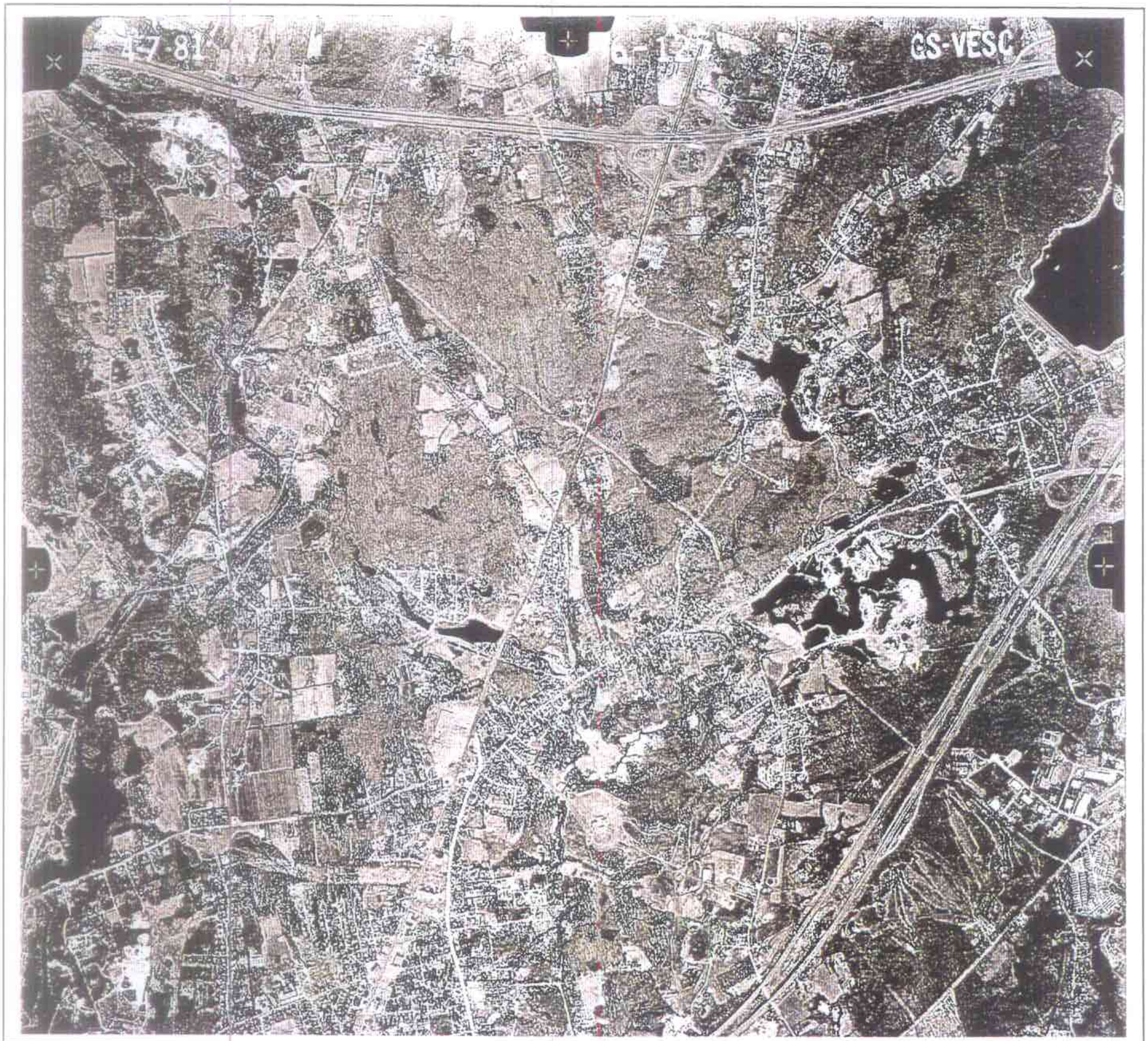

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APPENDIX D

DESCRIPTION OF THE SCREENING LEVEL WATERSHED MODEL

Description of Screening Level Lake Como Watershed Model

The landuse based spreadsheet model, developed by ENSR and applied to the Lake Como system, facilitates the development of a nutrient budget for Lake Como and the associated watershed. The model uses landuse specific runoff and export coefficients, and attenuation coefficients, to calculate the load of water and nutrients to Lake Como. In-lake nutrient concentrations, chlorophyll concentrations, and Secchi depths are then estimated, along with acceptable nutrient levels, to identify the need for a reduction in nutrient load. The use of the spreadsheet model progresses in a series of steps designed to allow the user to replicate the current watershed and in-lake situation and then develop alternative management scenarios to improve in-lake water quality. The spreadsheet modeling proceeds as follows.

1. *Delineate Watershed Boundaries* – The entire watershed is divided into several subbasins to isolate the areas of interest. Subbasins are generally developed to isolate specific landuse types or areas of contribution to specific storm water conduits. Once the subbasins have been delineated the surface areas of each is determined.

2. *Determination of Flow Using Runoff Coefficients* – Baseflow and stormwater runoff coefficients indicate the fraction of annual precipitation that flows out of the basin. Initial estimates of the baseflow and stormwater runoff coefficients are selected based on subbasin landuse type. Attenuation coefficients can be applied to the outflow to reduce the volume of water generated by the basin. The sum of the annual baseflow and stormflow for each subbasin describes the net annual outflow. Adjustments can be made to the baseflow and runoff coefficients if the model predicted outflow is not in close agreement with the standard water yield of 1.9 cfs/mi^2 (estimate for southern New England). Annual flow rates are calculated using the following equations.

$$\text{Baseflow } (m^3/yr) = \text{Precipitation } (m/yr) * \text{Baseflow Coefficient } (\%) * \text{Subbasin Area } (m^2)$$

$$\text{Stormwater flow } (m^3/yr) = \text{Precipitation } (m/yr) * \text{Runoff Coefficient } (\%) * \text{Subbasin Area } (m^2)$$

3. *Determination of Nutrient Loads Using Export Coefficients* – Landuse specific export coefficients for nitrogen and phosphorus are estimated to facilitate the calculation of nutrient loads from each subbasin. Initial estimates of landuse specific export coefficients are derived from Reckhow et. al. (1980). Attenuation coefficients can be applied to the nutrient loads to reduce the nutrient load generated by the basin. Adjustments can be made to the export coefficients to bring predicted baseflow and runoff nutrient concentrations into close agreement with measured values. Annual nutrient loads and concentrations are calculated using the following equations.

$$\text{Nutrient Load } (kg/yr) = \text{Export Coefficient } (kg/m^2/yr) * \text{Subbasin Area } (m^2)$$

$$\text{Nutrient Concentration } (kg/m^3) = \text{Nutrient Load } (kg/yr) / \text{Flow Rate } (m^3/yr)$$

4. *Calculation of In-Lake Phosphorus Concentrations* – Lake phosphorus concentrations are determined using the total subbasin phosphorus load, the physical geometry of the lake, and the flow through the lake. The maximum in-lake phosphorus concentration is determined using the mass balance approach or other methods including those of Kirchner and Dillon (1975), Vollenweider, (1975), Larsen and Mercier (1976), and Jones and Bachmann (1976), that incorporate flushing time, lake depth, and retention coefficients in their determination of in-lake phosphorus concentrations. During this phase of the modeling the predicted in-lake phosphorus concentrations can be compared with the measured concentrations to identify a need for adjusting either the runoff or exponent coefficients. The following five equations are used to estimate in-lake phosphorus concentrations.

   a) Mass Balance:

$$P\,Conc.(mg/L) = \frac{Total\ P\ Load\ (kg/m^2/yr)}{Depth\,(m)\cdot Flushing\ Rate\,(per\,Year)}$$

   b) Kirchner and Dillon, 1975:

$$P\,Conc.(mg/L) = \frac{Total\ P\ Load\ (kg/m^2/yr)\cdot(1-Settling\ \mathrm{Re}tention\,Coeff.)}{Depth\,(m)\cdot Flushing\ Rate\ (per\,Year)}$$

   c) Vollenweider, 1975:

$$P\,Conc.(mg/L) = \frac{Total\ P\ Load\ (kg/m^2/yr)}{Depth\,(m)\cdot(Suspended\ Fraction + Flushing\ Rate\ (per\,Year)}$$

   d) Larsen and Mercier, 1976:

$$P\,Conc.(mg/L) = \frac{Total\ P\ Load\ (kg/m^2/yr)\cdot(1-Flushing\ \mathrm{Re}tention\,Coeff.)}{Depth\,(m)\cdot Flushing\ Rate\ (per\,Year)}$$

   e) Jones and Bachmann, 1976:

$$P\,Conc.(mg/L) = \frac{0.84\cdot Total\ P\ Load\ (kg/m^2/yr)}{Depth\,(m)\cdot(0.65+Flushing\ Rate\ (per\,Year))}$$

5. *Calculation of Permissible/Critical Phosphorus Concentration* – In addition to the in-lake phosphorus concentrations, the permissible and critical phosphorus concentrations can be calculated using the depth and flushing rate. The permissible phosphorus concentration can be determined using the following equation by Vollenweider (1968). The critical concentration was simply a factor of 2 higher than the permissible. Permissible and critical concentrations are determined by multiplying the calculated phosphorus loads by an in-lake retention coefficient and dividing by the lake depth and flushing time.

    f) Vollenweider, 1968

$$Permissible\,(kg/m^2/yr) = \log^{-1}(0.501503 \cdot \log(Depth\,(m) \cdot Flushing\ Rate\,(per\,Year)) - 1.0018)$$

6. The development of predicted in-lake phosphorus concentrations and potential maximum target concentrations provides a means to quantify the effects of watershed management on in-lake phosphorus levels. Changes can be made to the runoff or export coefficients, attenuation coefficients, or the physical dimensions of a lake to reflect improvements resulting from watershed management alternatives.

The spreadsheet model developed by ENSR is a useful tool. Landuse data can be incorporated into the model to replicate nutrient concentrations measured throughout the watershed. Furthermore, the calculated watershed runoff and nutrient rates can be used to replicate in-lake nutrient concentrations. Once the model can effectively reproduce the current situation it can be easily adapted to other potential scenarios. In this way, alternative management practices can be tested to determine the viability of each alternative as a means of reducing nutrient concentrations and improving in-lake water quality.

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APPENDIX E

LAKE COMO MANAGEMENT FEASIBILITY ASSESSMENT

MANAGEMENT FEASIBILITY ASSESSMENT

EXISTING PROBLEMS

Obstacles to maximum environmental quality at Lake Como include:

- High levels of phosphorus, nitrogen, sediment, and other pollutants are present in storm water discharging to Lake Como.
- High levels of suspended solids in the water column in Lake Como.
- Low water levels during the summer in the main basin.
- Soft, nutrient-rich sediment promote growths of rooted plants, especially invasive, non-native species.

The primary impairments resulting from these problems include:

- Periodic algal blooms reduce water clarity.
- Resuspension and influxes of sediment reduces water clarity
- Rooted aquatic plant growths are excessive for designated uses.
- Aesthetically unappealing to lake users.

A complete management program will include watershed and in-lake components, as neither element alone appears sufficient to provide optimal long-term conditions in Lake Como

MANAGEMENT OPTIONS

Management options for Lake Como can be broken down into two broad categories, watershed management and in-lake management. Watershed management options will focus on pollutant loading in general, with particular emphasis on the control of phosphorus, the primary limiting nutrient for algal growth in this system. Most techniques are covered in more detail in several watershed management manuals (e.g., Schueler 1987, Dennis et al. 1989, Scheuler et al. 1992, Claytor and Scheuler 1996), but are summarized here for the purpose of evaluating applicability to the Lake Como watershed. In-lake management options will focus on algal control, rooted plant control, and possibly on mitigation of low levels of dissolved oxygen. The most detailed reference on this topic is by Cooke et al. (1993), but much material from a new book in the editorial stage has been used here as well. Many original citations are provided since that reference is not yet available.

Watershed Management Options

Source Reduction

Agricultural Best Management Practices

Agricultural Best Management Practices (BMPs) incorporate techniques in forestry, animal science, and crop science to minimize adverse impacts to water resources. This management approach actually relies upon a combination of techniques in source reduction and transport mitigation. Such practices include manure management, fertilizer management, use of cover crops, and use of buffer zones. The use of agricultural BMP's is not especially applicable in the Lake Como watershed, given the low percentage of agricultural lands.

Bank and Slope Stabilization

Erosion control is an important component of an overall management plan designed to decrease pollutant loading to aquatic ecosystems. This is especially important in areas of new development, where soils are both exposed and susceptible to erosion. Other critical areas include riparian zones and stream banks. This is a recommended management technique in the Lake Como watershed, as a matter of protection, and should be pursued vigorously in relation to new development.

Behavioral Modifications

Behavioral modifications involve changing the actions of watershed residents and lake users to improve water quality. Such changes may include conversion to non-phosphate detergents, limits on lawn fertilization, and eliminating illegal dumping in roadways and watercourses. Behavioral modifications can be brought about in two principal ways, through public education and/or the implementation of local bylaws and bans. Education is a critical first step and should precede any attempt at regulation.

Public education can be accomplished by mailing an informative brochure on watershed management to all residents in the watershed, through the use of video programs on local access television, by placing informative billboards in high access areas, or by holding public meetings for watershed residents. Public education relies heavily upon cooperation from residents and other lake users, and is not likely to result in major improvements in water quality by itself. However, some level of improvement has been noted in other studies and the education process sets the stage for community involvement and cooperation. Public education is a recommended management technique for Lake Como.

The focus of education and behavioral modifications in this watershed should be on fertilizer use and general storm water management for residential and commercial properties.

Land Use Conversion

Land use conversion involves purchasing properties that contribute excessive amounts of pollutants and converting these properties to less deleterious land uses. For example, the City may decide to purchase an industrial property and convert the land to open space, thus reducing pollutant generation from this parcel of land. This is a very expensive proposition and is not practical on a large scale in most case and appears to not be an option for the Lake Como watershed.

Storm Water Diversion

Re-routing a discharge away from a target water-body is one of the most effective ways to change the quality of incoming water. It suffers from the philosophical drawback of passing the problem downstream without dealing with the source of the pollution, and is not feasible in many areas where downstream uses must be protected. The model suggests that this option provides one of the highest phosphorus load reduction in the options presented in Section 3.6.4.2., however, redirecting stormwater drainage will reduce the amount of water entering the lake and it may take a long period of time to refill and/or maintain designated water levels.

Waste Water Management

A properly functioning on-site waste disposal system (e.g., septic system) can be an effective means of reducing pollutant loading to an aquatic ecosystem. Of particular concern are those systems where septic effluent is breaking-out above ground and is transported to the lake or a tributary during storm events. Also of concern are any systems located near or below the ground water table. Residences on the northern portion of the watershed are served by on-site waste disposal systems that could result in excessive nitrogen and phosphorus loading if systems are malfunctioning. This study did not reveal hard evidence of any malfunctions, but it is likely due the age of these systems they are likely not functioning optimally. Inputs are probably minor compared to the storm water load, but should be further evaluated. Hooking up this area to the sanitary sewer appears appropriate.

Zoning and Land Use Planning

This is a very important component in controlling watershed inputs to aquatic resources. A strong relationship exists between land use type and pollutant generation, with developed lands typically generating greater pollutant loads than non-developed lands. Preserving undeveloped land in the Lake Como watershed is highly recommended, but may not be completely practical. Choices between economic development and environmental quality must be made, unless stringent storm water controls are mandated. Planning should incorporate preservation of sensitive parcels, maintenance of buffer strips along waterways, and inclusion of major storm water controls. Effective controls often require 2-7% of the watershed area, which is no trivial land quantity in this case.

Transport Mitigation

Buffer Strips

Buffer strips (or vegetated filter strips or grassed buffers) are areas of grass or other dense vegetation that separate a waterway from an intensive land use. These vegetated strips allow overland flow to pass through vegetation that filters out some percentage of the particulates and decreases the velocity of the storm water. Particulate settling and infiltration of water often occurs as the stormwater passes through the vegetation. Buffer strips need to be at least 25 ft wide before any appreciable benefit is derived, and superior removal requires a width >100 ft. This can create land use conflicts, but creative planting and use of buffer strips can be a low cost, low impact means to minimize inputs to the aquatic environment.

This management technique is highly recommended for the Lake Como watershed. However, application is likely to be limited to new development and along the main basin. As a substantial portion of the pollutant load is associated with developed areas with drainage systems already in place, this approach is unlikely to yield the necessary load reductions by itself.

Catch Basins with Sumps and Hoods

Deep sump catch basins equipped with hooded outlets can be installed as part of a storm water conveyance system. Deep sumps provide capacity for sediment accumulation and hooded outlets prevent discharge of floatables (including non-aqueous phase hydrocarbons). Catch basins are usually installed as pre-treatment for other BMP's and are not generally considered adequate storm water treatment as a sole system. Volume and outlet configuration are key features which maximize particle capture, but it is rare that more than the coarsest fraction of the sediment/pollutant load is removed by these devices. This is a recommended management technique for the Lake Como watershed, but is not expected to be sufficient by itself to make an appreciable difference. Rather, this will be an important pre-treatment mechanism for infiltration strategies.

Chemical Treatment

In-stream chemical treatment involves the dosing of stream flows or stormwater discharges with alum or other coagulants to bind phosphorus and coagulate sediments to promote settling. During this process, phosphorus permanently complexes with aluminum or another binding agent, rendering it unavailable for biological uptake by algae. Clarification of drinking water uses alum extensively, and removal of phosphorus from waste water in tertiary treatment systems often involves alum. This in-stream/stormwater treatment technology has been successfully applied in other regions, especially Florida. A pilot application was performed on the primary tributary to a drinking water supply reservoir in Ohio, and another was conducted for the main inlet of a lake in Wellesley, MA, both with moderate success. The primary application of this technology has been for phosphorus removal where other BMPs were not viable. Phosphorus removal rates ranging from 50-95% have been reported. Removal rates ranging from 50-99% have also been documented for other pollutants such as suspended solids, nitrogen, color, and bacteria. Although not modeled, the result of this techniques would be greater than that achieved by Scenario 5 and 8 (use of infiltration chambers, fixing the dam and dredging the main basin, >41% decrease of phosphorus loading).

This may be a valid means of improving water quality in Lake Como, especially for controlling stormwater inputs. It carries a moderate capital cost and high operation and maintenance fees, but has potential for improving the lake on an as-needed basis. By treating during storms between about May and August, the summer water quality in the lake could be drastically improved. Careful considerations of dosing station placement and doses are needed. A pilot program could be initiated on a small scale to evaluate the efficacy of this techniques prior to the installation of something more permanent.

Wetlands

Wetlands are shallow pools that create conditions suitable for the growth of marsh or wetland plants. These systems maximize pollutant removal through vegetative filtration, nutrient uptake, soil binding, bacterial decomposition, and enhanced settling. Alternatively, a treatment system may combine created wetlands with detention ponds. Wetlands are suitable for on-line or off-line treatment (assuming maintenance of adequate hydrology with off-line systems to support the wetland). Allowing the upper basin to revert to a natural wetland or accelerating this process by dredging a meandering channel through the basin and allowing vegetation to replace some open water habitat would increase the attenuation of nutrients entering the main basin. The model predicted that phosphorus loading would decrease by 10% if allowed to revert naturally. This technique coupled with the use of infiltration chambers for stormwater drainage could reduce phosphorus loading by as much as 51%; increased attenuation (>51% P-load reduction) would be expected if aided by dredging and careful re-vegetation of the upper basin.

Detention

Detention ponds are essentially basins that are designed to hold a portion of storm water runoff for at least 12-24 hours. Pollutant removal is accomplished mainly through settling and biological uptake. Wet detention ponds are more effective than dry detention ponds as the latter have a greater risk of sediment re-suspension and generally do not provide adequate soluble pollutant removal. Although effective, the land requirement is typically large and available land in the key basins within the watershed of Lake Como is limited and does not appear to be a viable option at this time.

Infiltration Systems

Infiltration systems may include trenches, basins or dry wells, and involve the passage of water into the soil or an artificial medium. Particles are filtered by the soil matrix and many soluble compounds are adsorbed to soil particles. Such systems require sufficient storage capacity to permit the gradual infiltration of runoff. Pre-treatment of the runoff allows larger particles to be removed, thereby aiding in the prevention of infiltration system failure due to clogging and sediment accumulation.

Site constraints such as shallow depth to groundwater table or bedrock and poorly drained soils often limit the effective use of infiltration. In sites with suitable conditions, off-line infiltration systems are generally preferred. The key to successful infiltration is providing adequate pre-infiltration settling time or other treatment to remove particles that could clog the interface at which infiltration occurs. This is a recommended management technique for the Lake Como watershed in areas with appropriate soils and ground water elevations.

Each chosen site must be carefully evaluated for soil strata and permeability, much the way one would evaluate an area for a septic system. The key will be isolating the first flush, the portion with which most pollutants are associated, and providing adequate infiltration capacity over a reasonable time period (a day or two). As infiltration can occur in subsurface chambers, no major impact to surface uses is necessary.

Oil/Grit Chambers

A number of oil/grit chamber designs are currently on the market. These self-contained units include an initial settling chamber for sediment removal, typically have hooded internal passages to remove oil and other floatables, and often incorporate some form of outlet pool to control exit velocity. Several rely on a vortex design to enhance sediment removal (e.g., Vortechnics, Storm Defender). Such systems are most applicable as pre-treatment for other BMPs, and are generally well suited as retrofits for relatively small areas in developed watersheds. Installing these devices as off-line systems may enhance pollutant removal, but their more common use as on-line pre-treatment devices can be very beneficial. This is a recommended management technique for the Lake Como watershed in combination with infiltration technologies.

Street Sweeping/Catch Basin Cleaning

Removal of pollutants before they are washed into Lake Como could be accomplished by frequent street sweeping and catch basin cleaning. Both techniques provide only limited benefits by themselves, but could be effective tools in combination with other Best Management Practices. Truly effective street sweeping is accomplished with vacuum equipment, which costs in excess of $100,000/vehicular unit. Maintenance costs can also be substantial. Catch basin cleaning should be a semi-annual activity in any urban area, but rarely is; restoration of catch basin capacity is essential to the proper function of storm-water drainage systems, and costs about $50/catch basin per year when basins are cleaned on a bulk basis. Street sweeping and catch basin cleaning are recommended management techniques for the Lake Como watershed, as part of normal road maintenance and storm water drainage system management, but neither can be counted on as a primary pollutant control technique.

In-Lake Management Options – Algal Control

Excessive algal growth can become a serious nuisance in aquatic habitats. Two growth forms are most troublesome in lakes:

- Free-floating microscopic cells, colonies or filaments, called phytoplankton, that discolor the water and sometimes form green scum on the surface of the waterbody. These algae come from a variety of algal groups, including blue-greens, greens, diatoms, goldens, euglenoids and dinoflagellates, although the blue-greens tend to be the most troublesome group as a consequence of high densities, taste and odor issues, and possible toxins. These are the most likely problem algae in Lake Como. Although the resuspension of sediment and sediment loading are believe to be limiting algal growths in this system at this time. It is likely however, that decreasing sediment inputs and resuspension would increase light availability resulting in more frequent algal blooms.

- Mats of filamentous algae associated with sediments and weed beds, but often floating to the surface after a critical density is attained. These are most often green algae of the orders *Cladophorales* or *Zygnematales*, or blue-green algae (more properly cyanobacteria) of the order *Oscillatoriales*. There are some mats in Lake Como, but not at nuisance densities as far as we know.

Algae reproduce mainly through cell division, although resting cysts are an important mechanism for surviving unfavorable periods. When growth conditions are ideal (warm, lighted, nutrient-rich), algae multiply rapidly and reach very high densities (blooms) in a matter of days to weeks. Many algae contribute to taste and odor problems at high densities, and the decay of algae blooms can lead to oxygen depression.

The factors that control the abundance of algae form the basis for attempts to manage and limit them. Light and nutrients are the primary needs for algae growth. Where algal densities, non-algal turbidity, or shading by rooted plants do not create a light limitation, the quantity of algae in a lake is usually directly related to the concentration of the essential plant nutrient in least supply. In many cases this element is phosphorus. Even when phosphorus is not currently the limiting nutrient, it is usually more appropriate to create a phosphorus limitation than to control other nutrients. Algae management techniques (Table 1) such as dyes, artificial circulation and selective plantings seek to establish light limitation, while methods such as aeration, dilution and flushing, drawdown, dredging, phosphorus inactivation, and selective withdrawal are used to reduce nutrient availability.

TABLE 1. MANAGEMENT OPTIONS FOR CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
Physical Controls			
1) Hypolimnetic aeration or oxygenation	♦ Addition of air or oxygen at varying depth provides oxic conditions ♦ May maintain or break stratification ♦ Can also withdraw water, oxygenate, then replace	♦ Oxic conditions promote binding/sedimentation of phosphorus ♦ Counteraction of anoxia improves habitat for fish/invertebrates ♦ Build-up of dissolved iron, manganese, ammonia and phosphorus reduced.	♦ May disrupt thermal layers important to fish community ♦ May promote supersaturation with gases harmful to fish
2) Circulation and destratification	♦ Use of water or air to keep water in motion ♦ Intended to prevent or break stratification ♦ Generally driven by mechanical or pneumatic force	♦ Reduces surface build-up of algal scums ♦ Promotes uniform appearance ♦ Counteraction of anoxia improves habitat for fish/invertebrates ♦ Can eliminate localized problems without obvious impact on whole lake	♦ May spread localized impacts ♦ May increase oxygen demand at greater depths ♦ May promote downstream impacts
3) Dilution and flushing	♦ Addition of water of better quality can dilute nutrients ♦ Addition of water of similar or poorer quality flushes system to minimize algal build-up ♦ May have continuous or periodic additions	♦ Dilution reduces nutrient concentrations without altering load ♦ Flushing minimizes detention; response to pollutants may be reduced	♦ Diverts water from other uses ♦ Flushing may wash desirable zooplankton from lake ♦ Use of poorer quality water increases loads ♦ Possible downstream impacts

TABLE 1. MANAGEMENT OPTIONS FOR CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
4) Drawdown	♦ Lowering of water over autumn period allows oxidation, desiccation and compaction of sediments ♦ Duration of exposure and degree of dewatering of exposed areas are important ♦ Algae are affected mainly by reduction in available nutrients.	♦ May reduce available nutrients or nutrient ratios, affecting algal biomass and composition ♦ Opportunity for shoreline clean-up/structure repair ♦ Flood control utility ♦ May provide rooted plant control as well	♦ Possible impacts on contiguous emergent wetlands ♦ Possible effects on overwintering reptiles or amphibians ♦ Possible impairment of well production ♦ Reduction in potential water supply and fire fighting capacity ♦ Alteration of downstream flows ♦ Possible overwinter water level variation ♦ May result in greater nutrient availability if flushing inadequate
5) Dredging	♦ Sediment is physically removed by wet or dry excavation, with deposition in a containment area for dewatering ♦ Dredging can be applied on a limited basis, but is most often a major restructuring of a severely impacted system ♦ Nutrient reserves are removed and algal growth can be limited by nutrient availability	♦ Can control algae if internal recycling is main nutrient source ♦ Increases water depth ♦ Can reduce pollutant reserves ♦ Can reduce sediment oxygen demand ♦ Can improve spawning habitat for many fish species ♦ Allows complete renovation of aquatic ecosystem	♦ Temporarily removes benthic invertebrates ♦ May create turbidity ♦ May eliminate fish community (complete dry dredging only) ♦ Possible impacts from containment area discharge ♦ Possible impacts from dredged material disposal ♦ Interference with recreation or other uses during dredging
5.a) "Dry" excavation	♦ Lake drained or lowered to maximum extent practical ♦ Target material dried to maximum extent possible ♦ Conventional excavation equipment used to remove sediments	♦ Tends to facilitate a very thorough effort ♦ May allow drying of sediments prior to removal ♦ Allows use of less specialized equipment	♦ Eliminates most aquatic biota unless a portion left undrained ♦ Eliminates lake use during dredging

TABLE 1. MANAGEMENT OPTIONS FOR CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
5.b) "Wet" excavation	♦ Lake level may be lowered, but sediments not substantially exposed ♦ Draglines, bucket dredges, or long-reach backhoes used to remove sediment	♦ Requires least preparation time or effort, tends to be least cost dredging approach ♦ May allow use of easily acquired equipment ♦ May preserve aquatic biota	♦ Usually creates extreme turbidity ♦ Tends to result in sediment deposition in surrounding area ♦ Normally requires intermediate containment area to dry sediments prior to hauling ♦ May cause severe disruption of ecological function ♦ Usually eliminates most lake uses during dredging
5.c) Hydraulic removal	♦ Lake level not reduced ♦ Suction or cutterhead dredges create slurry which is hydraulically pumped to containment area ♦ Slurry is dewatered; sediment retained, water discharged	♦ Creates minimal turbidity and impact on biota ♦ Can allow some lake uses during dredging ♦ Allows removal with limited access or shoreline disturbance	♦ Often leaves some sediment behind ♦ Cannot handle coarse or debris-laden materials ♦ Requires sophisticated and more expensive containment area ♦ Requires overflow discharge from containment area
6) Light-limiting dyes and surface covers	♦ Creates light limitation	♦ Creates light limit on algal growth without high turbidity or great depth ♦ May achieve some control of rooted plants as well	♦ May cause thermal stratification in shallow ponds ♦ May facilitate anoxia at sediment interface with water
6.a) Dyes	♦ Water-soluble dye is mixed with lake water, thereby limiting light penetration and inhibiting algal growth ♦ Dyes remain in solution until washed out of system.	♦ Produces appealing color ♦ Creates illusion of greater depth	♦ May not control surface bloom-forming species ♦ May not control growth of shallow water algal mats
6.b) Surface covers	♦ Opaque sheet material applied to water surface	♦ Minimizes atmospheric and wildlife pollutant inputs	♦ Minimizes atmospheric gas exchange ♦ Limits recreational use

TABLE 1. MANAGEMENT OPTIONS FOR CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
7) Mechanical removal	♦ Filtering of pumped water for water supply purposes ♦ Collection of floating scums or mats with booms, nets, or other devices ♦ Continuous or multiple applications per year usually needed	♦ Algae and associated nutrients can be removed from system ♦ Surface collection can apply on an "as needed" basis ♦ May remove floating debris ♦ Collected algae dry to minimal volume	♦ Filtration requires high backwash and sludge handling capability for use with high algal densities ♦ Labor intensive unless a mechanized system applied, in which case it is capital intensive ♦ Many algal forms not amenable to collection by net or boom ♦ Possible impacts on non-targeted aquatic life
8) Selective withdrawal	♦ Discharge of bottom water which may contain (or be susceptible to) low oxygen and higher nutrient levels ♦ Intake of water from low algae layer to maximize supply quality ♦ May be pumped or utilize passive head differential	♦ Removes targeted water from lake efficiently ♦ Complements other techniques such as drawdown or aeration ♦ May prevent anoxia and phosphorus build up in bottom water ♦ May remove initial phase of algal blooms which start in deep water ♦ May create coldwater conditions downstream	♦ Possible downstream impacts of poor water quality ♦ May eliminate colder thermal layer important to certain fish ♦ May promote mixing of some remaining poor quality bottom water with surface waters ♦ May cause unintended drawdown if inflows do not match withdrawal
Chemical controls			
9) Algaecides	♦ Liquid or pelletized algaecides applied to target area ♦ Algae killed by direct toxicity or metabolic interference ♦ Typically requires application at least once/yr, often more frequently	♦ Rapid elimination of algae from water column , normally with increased water clarity ♦ May result in net movement of nutrients to bottom of lake	♦ Possible toxicity to non-target areas or species of plants/animals ♦ Restrictions on water use for varying time after treatment ♦ Increased oxygen demand and possible toxicity from decaying algae ♦ Possible recycling of nutrients, allowing other growths

TABLE 1. MANAGEMENT OPTIONS FOR CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
9.a) Forms of copper	♦ Contact algaecide ♦ Cellular toxicant, suggested disruption of photosynthesis, nitrogen metabolism, and membrane transport ♦ Applied as wide variety of liquid or granular formulations, often in conjunction with chelators, polymers, surfactants or herbicides	♦ Effective and rapid control of many algae species ♦ Approved for use in most water supplies	♦ Toxic to aquatic fauna as a function of concentration, formulation, temperature, pH, and ambient water chemistry ♦ Ineffective at colder temperatures ♦ Copper ion persistent; accumulates in sediments or moves downstream ♦ Certain green and bluegreen nuisance species are resistant to copper ♦ Lysing of cells releases cellular contents (including nutrients and toxins) into water column
9.b) Forms of endothall (7-oxabicyclo [2.2.1] heptane-2,3-dicarboxylic acid)	♦ Contact algaecide ♦ Membrane-active chemical which inhibits protein synthesis ♦ Causes structural deterioration ♦ Applied as liquid or granules, usually as hydrothol formulation for algae control	♦ Moderate control of thick algal mats, used where copper is ineffective ♦ Limited toxicity to fish at recommended dosages ♦ Rapid action	♦ Non-selective in treated area ♦ Toxic to aquatic fauna (varying degrees by formulation) ♦ Time delays on use for water supply, agriculture and recreation ♦ Safety hazards for applicators
9.c) Forms of diquat (6,7-dihydropyrido [1,2-2',1'-c] pyrazinediium dibromide)	♦ Contact algaecide ♦ Absorbed directly by cells ♦ Strong oxidant; disrupts most cellular functions ♦ Applied as a liquid, sometimes in conjunction with copper	♦ Moderate control of thick algal mats, used where copper alone is ineffective ♦ Limited toxicity to fish at recommended dosages ♦ Rapid action	♦ Non-selective in treated area ♦ Toxic to zooplankton at recommended dosage ♦ Inactivated by suspended particles; ineffective in muddy waters ♦ Time delays on use for water supply, agriculture and recreation

TABLE 1. MANAGEMENT OPTIONS FOR CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
10) Phosphorus inactivation	♦ Typically salts of aluminum, iron or calcium are added to the lake, as liquid or powder ♦ Phosphorus in the treated water column is complexed and settled to the bottom of the lake ♦ Phosphorus in upper sediment layer is complexed, reducing release from sediment ♦ Permanence of binding varies by binder in relation to redox potential and pH ♦ Potential for use on inlet streams as well	♦ Can provide rapid, major decrease in phosphorus concentration in water column ♦ Can minimize release of phosphorus from sediment ♦ May remove other nutrients and contaminants as well as phosphorus ♦ Flexible with regard to depth of application and speed of improvement	♦ Possible toxicity to fish and invertebrates, especially by aluminum at low pH ♦ Possible release of phosphorus under anoxia or extreme pH ♦ May cause fluctuations in water chemistry, especially pH, during treatment ♦ Possible resuspension of floc in shallow areas with extreme turbulence ♦ Adds to bottom sediment, but typically an insignificant amount
11) Sediment oxidation	♦ Addition of oxidants, binders and pH adjustors oxidizes sediment ♦ Binding of phosphorus is enhanced ♦ Denitrification is stimulated	♦ Can reduce phosphorus supply to algae ♦ Can alter N:P ratios in water column ♦ May decrease sediment oxygen demand	♦ Possible impacts on benthic biota ♦ Longevity of effects not well known ♦ Possible source of nitrogen for bluegreen algae
12) Settling agents	♦ Closely aligned with phosphorus inactivation, but can be used to reduce algae directly too ♦ Lime, alum or polymers applied, usually as a liquid or slurry ♦ Creates a floc with algae and other suspended particles ♦ Floc settles to bottom of lake ♦ Re-application typically necessary at least once/yr	♦ Removes algae and increases water clarity without lysing most cells ♦ Reduces nutrient recycling if floc sufficient ♦ Removes non-algal particles as well as algae ♦ May reduce dissolved phosphorus levels at the same time	♦ Possible impacts on aquatic fauna ♦ Possible fluctuations in water chemistry during treatment ♦ Resuspension of floc possible in shallow, well-mixed waters ♦ Promotes increased sediment accumulation

TABLE 1. MANAGEMENT OPTIONS FOR CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
13) Selective nutrient addition	♦ Ratio of nutrients changed by additions of selected nutrients ♦ Addition of non-limiting nutrients can change composition of algal community ♦ Processes such as settling and grazing can then reduce algal biomass (productivity can actually increase, but standing crop can decline)	♦ Can reduce algal levels where control of limiting nutrient not feasible ♦ Can promote non-nuisance forms of algae ♦ Can improve productivity of system without increased standing crop of algae	♦ May result in greater algal abundance through uncertain biological response ♦ May require frequent application to maintain desired ratios ♦ Possible downstream effects
14) Management for nutrient input reduction	♦ Generally not an in-lake process (See Chapter 6), but essential to note in any algal control program ♦ Includes wide range of watershed and lake edge activities intended to eliminate nutrient sources or reduce delivery to lake ♦ Can involve use of wetland treatment cells or detention areas created from part of lake ♦ Essential component of algal control strategy where internal recycling is not the dominant nutrient source, and desired even where internal recycling is important	♦ Acts against the original source of algal nutrition ♦ Decreased effective loading of nutrients to lake ♦ Creates sustainable limitation on algal growth ♦ May control delivery of other unwanted pollutants to lake ♦ Generally most cost effective over long term ♦ Facilitates ecosystem management approach which considers more than just algal control	♦ May involve considerable lag time before improvement observed ♦ May not be sufficient to acheive goals without some form of in-lake management ♦ Reduction of overall system fertility may impact fisheries ♦ May cause shift in nutrient ratios which favor less desirable species ♦ May cost more in the short term, as source management is generally more involved than one or a few treatments of symptoms of eutrophication

TABLE 1. MANAGEMENT OPTIONS FOR CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
Biological Controls			
15) Enhanced grazing	♦ Manipulation of biological components of system to achieve grazing control over algae ♦ Typically involves alteration of fish community to promote growth of large herbivorous zooplankton, or stocking with phytophagous fish	♦ May increase water clarity by changes in algal biomass or cell size distribution without reduction of nutrient levels ♦ Can convert unwanted biomass into desirable form (fish) ♦ Harnesses natural processes to produce desired conditions	♦ May involve introduction of exotic species ♦ Effects may not be controllable or lasting ♦ May foster shifts in algal composition to even less desirable forms
15.a) Herbivorous fish	♦ Stocking of fish which eat algae	♦ Converts algae directly into potentially harvestable fish ♦ Grazing pressure can be adjusted through stocking rate	♦ Typically requires introduction of non-native species ♦ Difficult to control over long term ♦ Smaller algal forms may be benefitted and bloom
15.b) Herbivorous zooplankton	♦ Reduction in planktivorous fish to promote grazing pressure by zooplankton ♦ May involve stocking piscivores or removing planktivores ♦ May also involve stocking zooplankton or establishing refugia	♦ Converts algae indirectly into harvestable fish ♦ Zooplankton community response to increasing algae can be rapid ♦ May be accomplished without introdu ction of non-native species ♦ Generally compatible with most fishery management goals	♦ Highly variable response expected; temporal and spatial variability may be problematic ♦ Requires careful monitoring and management action on 1-5 yr basis ♦ May involve non-native species introduction(s) ♦ Larger or toxic algal forms may be benefitted and bloom
16) Bottom-feeding fish removal	♦ Removes fish which browse among bottom deposits, releasing nutrients to the water column by physical agitation and excretion	♦ Reduces turbidity and nutrient additions from this source ♦ May restructure fish community in more desirable manner	♦ Targeted fish species are difficult to eradicate or control ♦ Reduction in fish populations valued by some lake users (human and non-human)

TABLE 1. MANAGEMENT OPTIONS FOR CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
17) Fungal/bacterial/viral pathogens	♦ Addition of inoculum to initiate attack on algal cells	♦ May create lakewide "epidemic" and reduction of algal biomass ♦ May provide sustained control for several years ♦ Can be highly specific to algal group or genera	♦ Largely experimental approach at this time ♦ Considerable uncertainty of results ♦ May promote resistant forms with high nuisance potential ♦ May cause high oxygen demand or release of toxins by lysed algal cells ♦ Effects on non-target organisms uncertain
18) Competition and allelopathy	♦ Plants may tie up sufficient nutrients to limit algal growth ♦ Plants may create a light limitation on algal growth ♦ Chemical inhibition of algae may occur through substances released by other organisms	♦ Harnesses power of natural biological interactions ♦ May provide responsive and prolonged control	♦ Some algal forms appear resistant ♦ Use of plants may lead to problems with vascular plants ♦ Use of plant material may cause depression of oxygen levels
18.a) Plantings for nutrient control	♦ Plant growths of sufficient density may limit algal access to nutrients ♦ Plants can exude allelopathic substances which inhibit algal growth	♦ Productivity and associated habitat value can remain high without algal blooms ♦ Portable plant "pods" , floating islands, or other structures can be managed to limit interference with recreation and provide habitat ♦ Wetland cells in or adjacent to the lake can minimize nutrient inputs	♦ Vascular plants may achieve nuisance densities ♦ There will be a water depth limitation on rooted plants but not algae ♦ Vascular plant senescence may release nutrients and cause algal blooms ♦ The switch from algae to vascular plant domination of a lake may cause unexpected or undesirable changes in lake ecology, especially energy flow
18.b) Plantings for light control	♦ Plant species with floating leaves can shade out many algal growths at elevated densities	♦ Vascular plants can be more easily harvested than most algae ♦ Many floating species provide valuable waterfowl food	♦ At the necessary density, the floating plants will be a recreational nuisance ♦ Low surface mixing and atmospheric contact will promote anoxia near the sediment

TABLE 1. MANAGEMENT OPTIONS FOR CONTROL OF ALGAE

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
18.c) Addition of barley straw	♦ Input of barely straw can set off a series of chemical reactions which limit algal growth ♦ Release of allelopathic chemicals can kill algae ♦ Release of humic substances can bind phosphorus	♦ Materials and application are relatively inexpensive ♦ Decline in algal abundance is more gradual than with algaecides, limiting oxygen demand and the release of cell contents	♦ Success appears linked to uncertain and potentially uncontrollable water chemistry factors ♦ Depression of oxygen levels may result ♦ Water chemistry may be altered in other ways unsuitable for non-target organisms ♦ Some forms of algae may be resistant and could benefit from the treatment

Natural algal losses occur through settling, consumption by grazers, and cellular death. Accelerated loss processes are the focus of techniques such as settling agents, biomanipulation (either grazing enhancement or addition of bacteria or viruses which kill algal cells), algaecide applications, and mechanical removal. Unfortunately, algae are remarkably adaptable as a community, and none of the techniques derived from loss processes are effective on the complete range of algae which commonly occur. Many blue-greens are buoyant and resist settling. Nuisance forms of green algae (*Chlorococcales* and *Cladophorales*) and certain blue-green algae (especially *Aphanizomenon*) are highly resistant to copper, the most common algaecide, and are also largely grazer-resistant. Only very dense algal mats can be feasibly harvested, and then with some difficulty.

Selective nutrient addition may provide an ecologically complex solution in some cases. By altering the ratio of nutrients, types of algae may be favored which are more amenable to other control techniques, most notably increased grazing pressure and settling rates. Productivity may not be reduced, but more efficient processing of primary production may lead to lower standing crop (biomass). Although sound in theory (Kilham 1971, Tilman 1982), this approach has rarely been applied in practical lake management efforts. It is important to recognize, however, that a productive lake need not suffer algal blooms (high biomass) if the algae and their consumers can be manipulated to increase the rate at which the energy represented by the algae can be passed to other trophic levels.

Filamentous algal mats have a distinctive ecology and are difficult to control. Mats typically form at the sediment-water interface or in association with rooted plant beds, taking nutrition from decay processes in that zone and surviving at low light levels through high densities of photosynthetic pigments. As mat density increases, photosynthetic gases are often trapped, and the mat may float upward and expand. Grazing control of mats is negligible, settling is not a major force, and harvesting is not practical in most cases. Algaecides are often ineffective once a dense mat has formed, as contact between algae and algaecide is limited. Prevention of mat formation through sediment removal or treatment (phosphorus inactivation or early algaecide application) is preferable to dealing with extensive, well-formed mats.

Many of the problems in potable water treatment are caused by eutrophic conditions in the water supply reservoir. Poor taste and odor are often associated with algal blooms and extensive benthic mats. Some common bloom-forming blue-green algae produce toxins lethal to domestic animals and may be linked to certain summer illnesses in humans. Combination of algal organic matter and chlorine added for disinfection can form disinifection byproducts such as trihalomethanes, which are potential carcinogens.

Use of algaecides typically releases taste and odor agents, toxins, other organic compounds, and nutrients into the water column, where they may remain a problem. Techniques that prevent the formation of high algal biomass are preferable to those that counteract the effects of a phytoplankton bloom or extensive mats. Where algaecides are to be used, maximum effectiveness is achieved by tracking algal composition and abundance and by timing treatments to coincide with the exponential growth phase of target algae.

Table 1 provides an overview of the techniques used to control algal abundance, with notes on the mode of action, advantages, and disadvantages of each technique. Additional details are provided in narrative form below.

Aeration or Oxygenation

Aeration puts air into the aquatic system, increasing oxygen concentration by transfer from gas to liquid and generating a controlled mixing force. The oxygen transfer function is most appropriately used to prevent hypolimnetic anoxia. By keeping the hypolimnion from becoming anoxic during stratification, aeration should minimize the release of phosphorus, iron, manganese and sulfides from deep bottom sediments and decrease the build-up of undecomposed organic matter and compounds (e.g., ammonium). Hypolimnetic aeration can also increase the volume of water suitable for habitation by zooplankton and fish, especially coldwater forms. Pure oxygen can be used in place of air to maximize transfer. Permits are generally required for aeration projects, but hypolimnetic aeration is among the easier lake management processes to get approved, having few adverse side effects.

Aeration is commonly used to mix shallow lakes, and is sometimes used as a mixing force to destratify deep lakes. Aeration in an extremely shallow water body could resuspend more soft sediment and is not recommended in Lake Como at this time, especially if the impoundment is not sufficiently dredged.

Circulation and Destratification

Circulation affects mixing and the uniformity of lake conditions. Thermal stratification and features of lake morphometry such as coves create stagnant zones which may be subject to loss of oxygen, accumulation of sediment, or algal blooms. Artificial circulation minimizes stagnation and can eliminate thermal stratification or prevent its formation. Movement of air or water is normally used to create the desired circulation pattern in shallow (<20 ft) lakes. Surface aerators, bottom diffusers, and water pumps have all been used to mix small ponds and shallow lakes. The effect is largely cosmetic in many instances; algae are simply mixed more evenly in the available volume of water.

Stratification is broken or prevented in deeper lakes through the injection of compressed air into lake water from a diffuser at the lake bottom. The rising column of bubbles, if sufficiently powered, will produce lakewide mixing at a rate that eliminates temperature differences between top and bottom waters. The use of air as the mixing force also provides some oxygenation of the water, but the efficiency and magnitude of this transfer are generally low. In some instances, wind driven pumps have been used to move water.

This technique is not applicable to Lake Como, as the lake does not stratify during the summer months.

Dilution and Flushing

Lake waters that have low concentrations of an essential nutrient are unlikely to exhibit algal blooms. While it is preferable to reduce nutrient loads to the lake, it is possible to lower (dilute) the concentration of nutrients within the lake by adding sufficient quantities of nutrient-poor water from some additional source. High amounts of additional water, whether low in nutrients or not, can also be used to flush algae out of the lake faster than they can reproduce. However, complete flushing is virtually impossible in many lake systems; small, linear impoundments are the primary candidates for such treatment.

Dilution or flushing washes out algal cells, but since the reproductive rate for algae is high (blooms form within days to a few weeks), only extremely high flushing rates will be effective. A flushing rate of 10 to 15% of the lake volume per day is appropriate. Development of reliable water and nutrient budgets are necessary to an evaluation of flushing as an algae control technique.

Very few documented case histories of dilution or flushing exist, in part because additional water is not often available, especially water that is low in nutrients. The best documented case of dilution is that of Moses Lake, Washington (Welch and Patmont, 1980; Cooke et al. 1993a), where low-nutrient Columbia River water was diverted through the lake. Water exchange rates of 10 to 20% per day were achieved, algal blooms dramatically decreased, and transparency was markedly improved, illustrating the potential effectiveness of this method.

Outlet structures and downstream channels must be capable of handling the added discharge for this approach to be feasible. Qualitative downstream impacts must also be considered. Water used for dilution or flushing should be carefully monitored prior to use in the lake.

It is not clear where enough water of any quality could be obtained to dilute or flush Lake Como, so this technique is not favored in this case.

Drawdown

By lowering the water level and exposing sediments, those sediments can be oxidized and compacted. This is expected to lower the oxygen demand and long-term phosphorus release rate from those sediments. Recent research (Mitchell and Baldwin 1998) indicates that shifts in the bacterial community during exposure may be responsible for reduced phosphorus release after some drawdowns. While the theory is attractive, in practice this approach suffers from several important limitations:

- The most problematic sediments are in the deepest part of most lakes, necessitating a complete draining of the lake for maximum effect
- Dewatering the sediments to the extent necessary to get more than minor surficial oxidation and compaction is difficult

- Nutrient release upon refill of the lake may actually increase until the nutrients released from organic decay can be flushed from the system
- The ability to control drawdown is dependent upon the presence of a manageable outlet structure and system hydrology, features normally associated only with impoundments.

Cooke et al. (1993a) reports on a variety of lakes which were subjected to drawdown, with variable results, and describes some of the factors which may control the effectiveness of this technique for algal control. It is apparent that little damage is done to algal resting cysts by drawdown, and that reduced algal densities are a function of reduced phosphorus availability. Chemical and physical features of the sediments influence the impact of drawdown, and the effects of this technique are easily overwhelmed by elevated external phosphorus loads. Use of this technique for algal control appears uncommon and unreliable, and is not recommended for Lake Como.

Dredging

The release of algae-stimulating nutrients from lake sediments can be controlled by removing layers of enriched materials. This may produce significantly lower in-lake nutrient concentrations and less algal production, assuming that there has been adequate diversion or treatment of incoming nutrient, organic and sediment loads from external sources. Even where incoming nutrient loads are high, dredging can reduce benthic mat formation and related problems with filamentous green and blue-green algae, as these forms may initially depend on nutrient-rich substrates for nutrition. Dredging also removes the accumulated resting cysts deposited by a variety of algae. Although recolonization would be expected to be rapid, changes in algal composition can result.

Sediment removal to retard nutrient release can be effective. An example is provided by Lake Trummen in Sweden (Andersson 1988) where the upper 3.3 feet of sediments were extremely rich in nutrients. This layer was removed and the total phosphorus concentration in the lake dropped sharply and remained fairly stable for at least 18 years. Phytoplankton production was reduced as a result.

Algal abundance also decreased and water clarity increased in Hills Pond in Massachusetts after all soft sediment was removed and a storm water treatment wetland was installed in 1994 (Wagner 1996). Dredging of 6-acre Bulloughs Pond in Massachusetts in 1993 has resulted in abatement of thick green algal mats for five years now, despite continued high nutrient loading from urban runoff (Wagner, pers. obs.). These mats had previously begun as spring bottom growths, then floated to the surface in mid-summer.

While removing the entire nutrient-rich layer of sediment can control algae, dredging is most frequently done to deepen a lake, remove accumulations of toxic substances, or to remove and control macrophytes. Algal control benefits are largely ancillary in these cases. The expense of complete soft sediment removal and the more pressing need for watershed management in most cases are the primary reasons that dredging is not used more often for algal control. Dredging is recommend for Lake Como, but necessarily based on algal control. Dredging would increase water depth, increase aesthetics, increase recreational opportunities, remove unwanted rooted plants and remove any internal phosphorus loading that may be occurring in this system.

Light Limitation by Dyes and Surface Covers

Dyes are sometimes lumped with algaecides in discussions of management options and are often subject to the same permit process as algaecides. However, their mode of action is to inhibit light penetration and resultant photosynthesis, so they are more properly classified as a physical technique. Dyes are intended to be inert pigments (typically blue) that produce a pronounced but generally aesthetic color in the water column. No direct toxic effects have been reported, and organisms in the water are not subject to coloration. Visibility is reduced as a function of limited light penetration, in proportion to the concentration of dye achieved. Under light limitation, algal production is expected to be reduced.

Dyes can be effective under certain circumstances, where algal production over substantial water depth is causing problems, but growths in shallow waters are unlikely to be significantly reduced. Dyes will not always eliminate floating scums or mats, and may actually promote surface growths. Combined with a circulation system, dyes can mask otherwise unpleasant algal blooms and improve the aesthetic appeal of ponds, reflecting pools, or similar waterbodies.

Repetitive treatment is necessary as the dye is flushed from the system, but treatments are relatively inexpensive. The greatest threat of negative impact involves the establishment of thermal stratification in as little as 6 to 8 ft of water as a consequence of limited light penetration. Anoxia at the sediment-water interface may ensue, creating the potential for a variety of impacts on water quality.

Surface covers are opaque sheets which, when placed on the surface of a lake or reservoir, inhibit light penetration. This technique creates access limitations for recreational lakes, but has been used in reservoirs, especially for storage of “finished” water (treated water ready for distribution). Aside from minimizing algal growth, such covers aid compliance with the Federal Safe Drinking Water Act by limiting interaction with waterfowl and atmospheric deposition. Most covered storage reservoirs have a more structural cover, such as a roof, but plastic covers are used in some cases and seem to perform acceptably.

Given the conditions in Lake Como and the desire for greater water clarity, not color, dyes do not appear appropriate in this case. Furthermore, dyes do not address the root cause of the problem, elevated nutrient levels. Techniques are available to control nutrients, so a more cosmetic approach is not recommended at this time.

Mechanical Removal
Algae are routinely removed from reservoir water by mechanical means in treatment facilities which provide drinking water to the public, but such sedimentation or filtration systems have not been developed for use with strictly recreational lakes. Treating enough lake water to cause an overall reduction in algal levels in a well mixed lake would be very difficult, and is likely to be cost prohibitive. McComas (1993) reviews methods which might be suitable for small recreational lakes, but generally concludes that this is not a cost effective approach.

It is also possible to harvest algal mats with nets, booms or commercial macrophyte harvesting equipment, but this can also be very expensive on a lakewide basis over the course of a summer. Collection with nets or a boom system may provide temporary improvement in small ponds, but is likely to be inefficient even then. Collection with harvesting equipment on a larger scale requires frequent offloading per unit of dry weight collected, since algae are mostly water.

The extent of matted algae present in this system does not warrant the use of mechanical harvesting.

Selective Withdrawal
For recreational lake management, the intent of selective withdrawal is usually to remove the poorest quality water from the lake, which is normally the water at the bottom of the lake unless an intense surface bloom of algae is underway. It is desirable to discharge water at a rate that prevents anoxia near the sediment-water interface, resulting in both improved lake conditions and an acceptable discharge quality. This can be accomplished in impoundments with small hypolimnia and/or large inflows. In most lake management cases, however, selective withdrawal will involve waters of poor quality and treatment may be necessary before discharge downstream.

This technique is not appropriate for a shallow lake like Lake Como and is not recommended.

Algaecides
Algaecides are toxic substances directed at algae to reduce their abundance in the water column or at the sediment-water interface. The oldest and still most used algaecide is copper, although it comes in a wide variety of forms, some of which are new formulations. Copper is a cellular toxicant (Westerdahl and Getsinger 1988a) and is believed to inhibit algal photosynthesis, alter nitrogen metabolism, and disrupt membrane transport functions. Copper sulfate (CuSO4) is the most common and basic form, with various chelators and/or surfactants added in other formulations to prolong or enhance effectiveness. It is registered for use in potable waters, although restrictions apply in most states.

Copper sulfate can be applied by towing burlap or nylon bags filled with granules (which dissolve) behind a boat. Other formulations can be applied as broadcast granules or sprayed liquids. A copper slurry can be delivered to an intended depth by a weighted hose. The method of delivery is not as important as the duration of effectiveness, however. In alkaline waters (150 mg calcium carbonate per liter, or more) or in waters high in hardness or organic matter, copper can be quickly lost from solution and thus rendered ineffective. In these cases, a liquid chelated form is often used. This formulation allows the copper to remain dissolved in the water long enough to kill algae. Dilution is another important factor, as copper is often applied to only the upper 10 ft of water to provide a deeper refuge for zooplankton and sensitive fish species. Vertical or horizontal mixing can rapidly decrease doses below an effective level. A review of dose effectiveness and environmental impacts is found in Cooke and Carlson (1989).

Most planktonic algae and periphytic algae forming loose filamentous mats in weed beds or on the bottom will be killed by doses of 1-2 mg $CuSO_4$/L (0.4 to 0.8 mg Cu/L). In most cases, cells lyse and release their contents into the water column. Copper sulfate is often effective against many green and blue-green algae, and against nearly all diatoms, golden algae, dinoflagellates, cryptomonads, and euglenoids, although the response may be brief and sustained control may require additional applications. However, some planktonic forms appear resistant to copper, including the filamentous blue-green alga *Aphanizomenon* and many species of the green algal order *Chlorococcales*. Strains of the filamentous blue-greens *Anabaena, Oscillatoria and Phormidium* have exhibited high resistance to copper in some reservoirs. Additionally, dense algal mats, especially those formed from members of the *Cladophorales*, are resistant by virtue of the inability of copper to come in contact with more than the outer layer of filaments. As these are some of the most severe nuisance forms, copper treatments may eventually cause greater algal problems by favoring resistant species.

An additional concern with the use of algaecides is the release of toxins common to certain strains of some species of blue-greens (Kenefick et al. 1993). Killing the cells releases the toxins into the water, where they may persist for an undetermined period. While the existence of these toxins has been known for many decades, recent improvement in detection levels has revealed more widespread occurrence and sublethal effects (Kotak et al. 1993). Although the use of activated carbon in water treatment removes these toxins, simple filtration does not. No human fatalities have been documented, but human illnesses appear linked to these toxins, and related deaths of wildlife and domestic animals have been confirmed (Haynes 1988).

The toxicity of copper to lake fauna presents a major risk of food web perturbation from copper treatments. Cooke et al. (1993a) review the literature on copper toxicity in lakes and conclude that acute toxicity to fish is possible at normal copper doses, although sublethal effects appear more likely. Zooplankton species are especially sensitive to copper, with reproductive impairment or mortality at concentrations 10 to 100 times lower than commonly applied doses. Loss of zooplankton affects both grazing control of algae and food resources for many fish species. Benthic invertebrates have also been found to be sensitive to copper within the normally applied range.

Hanson and Stefan (1984) suggest that 58 years of copper sulfate use in a group of Minnesota lakes, while effective at times for the temporary control of algae, appears to have produced dissolved oxygen depletions, increased internal nutrient cycling, occasional fishkills, copper accumulation in sediments, increased tolerance to copper by some nuisance blue-green algae, and undesirable impacts on fish and zooplankton. Short-term control (days) of algae may have been traded for long-term degradation of the lakes, but conclusive studies are rare.

Alternatives to copper-based algaecides are few. Endothall (as the hydrothol formulation) and diquat are still used with some success against hard-to-kill greens and blue-greens, but water use is restricted for at least a week after application, and diquat can be toxic to many lake invertebrates. Use in drinking water supplies is prohibited in Connecticut. New formulations of copper are more common than new non-copper-based algaecides.

Given the many negative aspects of algaecide applications, especially those involving copper, such treatments should only be used as the last line of defense. Frequent need for algaecides should be taken as an indication that a more comprehensive management plan is needed. Where algaecides are used, effectiveness is enhanced through improved timing of application. All too often algaecides are applied after a bloom has formed, instead of early in the exponential growth phase, when algal sensitivity is greatest and the impacts of lysing cells on the aquatic environment are minimized. Proper timing of application requires daily to weekly tracking of algal populations, potentially at greater annual expense than the actual annual treatment cost. This is not a valid, long-term management approach for Lake Como.

Phosphorus Inactivation

The release of phosphorus stored in lake sediments can be so extensive in some lakes and reservoirs that algal blooms persist even after incoming phosphorus has been significantly lowered. Phosphorus precipitation by chemical complexing removes phosphorus from the water column and can control algal abundance until the phosphorus supply is replenished. Phosphorus inactivation typically involves some amount of phosphorus precipitation, but aims to achieve long-term control of phosphorus release from lake sediments by adding as much phosphorus binder to the lake as possible within the limits dictated by environmental safety. It is essentially an “anti-fertilizer” addition. This technique is most effective after nutrient loading from the watershed is sufficiently reduced, as it acts only on existing phosphorus reserves, not new ones added post-treatment.

Aluminum has been widely used for phosphorus inactivation, mostly as aluminum sulfate and sometimes as sodium aluminate, as it binds phosphorus well under a wide range of conditions, including anoxia. However, concentrations of reactive aluminum (AL^{+3}) are strongly influenced by pH, and levels in excess of 50 ug/l may be toxic to aquatic fauna. A pH of between 6.0 and 8.5 virtually ensures that the 50 ug/l limit will not be reached, but aluminum sulfate addition can reduce the pH well below a pH of 6.0 in poorly buffered waters. In such cases sodium aluminate, which raises the pH, has been successfully used in combination with aluminum sulfate (Cooke et al. 1993b). It is also possible to add buffering agents to the lake prior to aluminum sulfate addition, such as lime and sodium hydroxide. Other chemicals that have been successfully employed to bind phosphorus include calcium hydroxide and ferric chloride; the former tends to raise the pH and the latter lowers the pH slightly. Ferric sulfate has also been applied, and lowers the pH substantially.

In practice, aluminum sulfate (often called alum) is added to the water and colloidal aggregates of aluminum hydroxide are formed. These aggregates rapidly grow into a visible, brownish white floc, a precipitate that settles to the sediments in a few hours to a few days, carrying sorbed phosphorus and bits of organic and inorganic particulate matter in the floc. After the floc settles to the sediment surface, the water will be very clear. If enough alum is added, a layer of 1 to 2 inches of aluminum hydroxide will cover the sediments and significantly retard the release of phosphorus into the water column as an internal load. In lakes where sufficient reduction of external nutrient loading has occurred, this can create a phosphorus limitation on algal growth.

Good candidate lakes for this procedure are those that have had external nutrient loads reduced to an acceptable level and have been shown, during the diagnostic-feasibility study, to have a high internal phosphorus load (release from sediment). High alkalinity is also desirable to provide buffering capacity. Highly flushed impoundments are usually not good candidates because of an inability to limit phosphorus inputs. Treatment of lakes with low doses of alum may effectively remove phosphorus from the water column, but may be inadequate to provide long-term control of phosphorus release from lake sediments.

Nutrient inactivation has received increasing attention over the last decade as long lasting results have been demonstrated in multiple projects, especially those employing aluminum compounds (Welch and Cooke 1999). Annabessacook Lake in Maine suffered algal blooms for 40 years prior to the 1978 treatment with aluminum sulfate and sodium aluminate (Cooke et al. 1993a). Low buffering capacity necessitated the use of sodium aluminate. A 65% decrease in internal phosphorus loading was achieved, blue-green algae blooms were eliminated, and conditions have remained much improved for nearly 20 years. Similarly impressive results have been obtained in two other Maine Lakes using the two aluminum compounds together (Connor and Martin 1989a).

Kezar Lake was treated with aluminum sulfate and sodium aluminate in 1984 after a wastewater treatment facility discharge was diverted from the lake. Both algal blooms and oxygen demand were depressed for several years, but began to rise more quickly than expected (Connor and Martin 1989a, 1989b). Additional controls on external loads (wetland treatment of inflow) reversed this trend and conditions have remained markedly improved over pre-treatment conditions for almost 15 years. No adverse impacts on fish or benthic fauna have been observed despite careful monitoring.

Aluminum sulfate and sodium aluminate were again employed with great success at Lake Morey, Vermont (Smeltzer 1990). A pretreatment average spring total phosphorus concentration of 37 ug/l was reduced to 9 ug/l after treatment in late spring of 1987. Although epilimnetic phosphorus levels have varied since then, the pretreatment levels have not yet been approached. Hypolimnetic phosphorus concentrations have not exceeded 50 ug/l. Oxygen levels increased below the epilimnion, with as much as 10 vertical feet of suitable trout habitat reclaimed. Some adverse effects of the treatment on benthic invertebrates and yellow perch were suggested to be temporary phenomena following treatment.

Phosphorus inactivation has also been successful in some shallow lakes (Welch et al. 1988, Gibbons 1992, Welch and Schrieve 1994), but have been unsuccessful in cases where the external loads have not been controlled prior to inactivation (Barko et al. 1990, Welch and Cooke 1999). Successful dose rates have ranged from 3 to 30 g Al/m^3 (15 to 50 g Al/m^2) with pH levels remaining above 6.0. Jar tests are used to evaluate the appropriate dose. A ratio of aluminum sulfate to sodium aluminate of 2:1 is expected to cause no change in system pH. Maintenance of the ambient pH is an appropriate goal, unless the pH is especially high as a consequence of excessive algal photosynthesis.

Aluminum sulfate is often applied near the thermocline depth (even before stratification) in deep lakes, providing a precautionary refuge for fish and zooplankton that could be affected by dissolved reactive aluminum. Application methods include modified harvesting equipment, outfitted pontoon boats, and specially designed barges made for this purpose.

Success has also been achieved with calcium (Babin et al. 1989, Murphy et al. 1990) and iron (Walker et al. 1989) salts, but it has become clear that aluminum provides the greatest long-term binding potential for phosphorus inactivation (Harper et al. 1999). The use of calcium would seem to be appropriate in high pH lakes, and provides natural phosphorus inactivation in certain hardwater lakes. Iron seems to be most useful in conjunction with aeration systems. Aluminum salts can be used successfully in any of these cases, however, and alum tends to be the chemical of choice unless toxicity becomes a problem.

Longevity of alum treatments has generally been excellent where external inputs of phosphorus to the system have been controlled (Payne et al. 1991). As a general rule, inactivation with aluminum can be expected to last for at least three flushing cycles, with much longer effectiveness where external loading has been controlled. A review of 21 well-studied phosphorus inactivation treatments using aluminum (Welch and Cooke 1999) indicates that longevity of effects is typically 15 years or more for dimictic (summer stratified) lakes and about 10 years for shallow, polymictic (unstratified) lakes.

Despite major successes, addition of aluminum salts to lakes does have the potential for serious negative impacts, and care must therefore be exercised with regard to dosage and buffering capacity. The potential for toxicity problems is directly related to the alkalinity and pH of the lake water. In soft (low alkalinity) water, only very small doses of alum can be added before alkalinity is exhausted and the pH falls below 6.0. At pH 6.0 and below, $Al(OH)_2$ and dissolved elemental aluminum (Al^{+3}) become the dominant forms. Both can be toxic to aquatic species. Well-buffered, hard water lakes can handle much higher alum doses without fear of creating toxic forms of aluminum. Soft water lakes must be buffered, either with sodium aluminate or other compounds, to prevent the undesirable pH shift while allowing enough $Al(OH)_3$ to be formed to control phosphorus release.

Although pH depression is the major threat, elevated pH from over-buffering can also cause problems. Hamblin Pond in Massachusetts was treated with alum and sodium aluminate in 1995, after three years of pre-treatment study which demonstrated both the importance of internal phosphorus loading and limited buffering capacity (Wagner 1999). A number of problems arose during the treatment, resulting in an overdose of sodium aluminate throughout the lake. The pH rose from about 6.3 to over 9.0, and a fish kill resulted. Despite an increase in summer water transparency from about 4 ft to nearly 20 ft and a gain of 10 vertical feet of suitable coldwater fish habitat, the fish kill has fostered some sentiments against this technique.

Other potential adverse impacts relate to the spread of macrophytes and changes in water chemistry after addition of aluminum compounds. Although the sharp increase in water transparency is viewed as desirable in most cases, it may allow an existing rooted plant infestation to spread into new areas or deeper water. The addition of sulfates to the lake in an aluminum sulfate treatment may foster chemical reactions that disrupt the iron cycle and associated natural phosphorus binding capacity. Aluminum sulfate treatments that reduce the pH may cause decalcification in sensitive organisms and may also limit calcium control of phosphorus cycling. Aluminum toxicity to humans has created substantial public controversy as regards treatment of lakes with aluminum, but concerns have not been supported by the bulk of scientific investigations (Krishnan 1988, Harriger and Steelhammer 1989). A detailed knowledge of lake chemistry is necessary to understand and apply phosphorus inactivation as an algal control technique.

In lake nutrient inactivation would be not be cost effective given the high watershed nutrient load to the lake. This technique could be re-evaluated once watershed, specifically stormwater, inputs are controlled.

Sediment Oxidation

The goal of this procedure is to decrease phosphorus release from sediments, as with aeration, drawdown, or phosphorus inactivation. If sediments are low in iron, ferric chloride or similar compounds can be added to enhance phosphorus binding. Lime is also added to raise sediment pH to 7.0-7.5, the optimum pH for denitrification. Then calcium nitrate is injected into the top 10 inches of sediments to promote the oxidation or breakdown of organic matter and denitrification. The entire procedure is often called Riplox after its originator, W. Ripl.

Lake Lillesjon, a 10.5-acre Swedish lake with a 6.6-foot mean depth, was the first to be treated (Ripl 1976). The treatment lowered sediment phosphorus release dramatically and lasted at least two years. A portion of a Minnesota lake was also treated, but high external loading overwhelmed the effects. No negative impacts have been reported, but the impact on benthic communities could be severe. However, where this technique is appropriate, there will probably not be a significant benthic community prior to treatment.

Although developed in the 1970s, this technique is not widely used and is in need of further experimentation. Oxidation and other reactions that alter sediment chemistry would seem to have great potential for controlling internal loads of a variety of contaminants.

Settling Agents

The water treatment industry has a long history of coagulant use intended to enhance settling and filtration. Application of such settling agents in lakes is theoretically possible, but examples include mainly alum that is applied as much for phosphorus inactivation and indirect algae control as for direct removal of algae. Calcium compounds have been used in several Alberta lakes (Babin et al. 1989, Murphy et al. 1990, Prepas et al. 1990), again as much for phosphorus inactivation as for direct algal removal, but such treatments do cause most algae to settle to the bottom. Various polymers could be used, but there is little documentation of this approach.

The primary value of this technique over algaecides is its ability to remove algal cells from the water column, rather than lysing cells in the water column and releasing their contents throughout the lake. Settling may eventually result in the release of cellular contents, but not rapidly and not throughout the water column. Not all algal species are amenable to such settling, however; many blue-green species with buoyancy vacuoles resist settling unless a strong floc layer develops and sweeps them out of the water column. Underdosing in such cases may not reduce algal densities to the extent expected, or may result in the formation of unsightly macroscopic clumps.

While potentially feasible in Lake Como, this approach is not preferable to nutrient controls.

Selective Nutrient Addition

This is another approach with theoretical appeal that has not been subjected to widespread practical study. In theory, a change in nutrient ratios should drive a shift in algal composition as a consequence of competitive superiority by species better suited to the new ratio (Kilham 1971, Tilman 1982). If the shift in algal composition results in dominance by algae of lower nuisance potential, or species which can be more readily consumed by zooplankton, the standing crop of algal biomass might be reduced or at least distributed in a way which would improve human perception of lake condition. Laboratory and some whole lake experiments have supported this theory, but well documented practical applications are lacking.

In reality, competitive forces seem weak compared to predation and grazing pressure, and changes in environmental conditions other than nutrient ratios would be expected to create instability that limits competitive effects. While this approach may be appropriate under certain circumstances, it does not appear likely to become a central technique in lake management and is not appropriate for Lake Como.

An alternative nutrient input strategy involves the addition of nitrate to a hypolimnion to serve as an alternative electron acceptor (Kortmann and Rich 1994). This would limit generation of sulfides from sulfates, reduce iron-sulfide reactions, and enhance iron-phosphorus binding. Anaerobic hypolimnetic metabolism would act to release the nitrogen as a gas, minimizing any uptake by algae. The nitrogen could be added as aluminum nitrate or ferric nitrate, further enhancing phosphorus binding activity. It may also be possible to utilize cold ground water high in nitrates for this purpose. As with alteration of the epilimnetic nutrient balance, this approach is theoretically sound but in need of well documented practical applications. This is nor a preferred method for Lake Como as watershed controls are more important than any temporary relief gained by this in lake technique.

Management for Nutrient Input Reduction

Techniques that belong to this category are largely watershed management methods discussed previously. The boundary blurs, however, when a part of the lake is used to create a detention, filtration or wetland treatment area to improve the quality of incoming water. The distinction is also less clear when tributary or storm water runoff is treated with phosphorus inactivators prior to discharge to a lake, where the floc then settles (Harper et al. 1999). However, it is important to recognize that at least some portion of the pollutant attenuation function ascribed to watersheds can be assumed by part of the lake, with proper planning and implementation. Nutrient input reductions for Lake Como should focus on the watershed before further consideration is given to devoting part of the lake to engineered treatment processes. However, in-lake management may compare favorably to watershed controls on economic grounds, at least over a period of several decades, and may be appropriate as a shorter term method to achieve water quality goals until watershed controls can be more fully implemented.

Enhanced Grazing

Shapiro et al. (1975) suggested a group of procedures, called "biomanipulation," that they believed could greatly improve lake quality without the use of expensive machines or chemicals. Much like the selective nutrient addition described previously, this technique depends upon general ecological principles to manipulate biological components of the lake or reservoir to produce desired conditions. In this regard, grazing is viewed as a potentially powerful force in structuring the algal community. Unlike selective nutrient addition, however, biomanipulation for grazing enhancement has been performed in many systems, often with satisfactory results.

In some lakes the amount of algae in the open water is controlled at times by grazing zooplankton rather than by the quantity of nutrients (McQueen et al. 1986b). Productivity may be high, but grazing prevents the produced biomass from accumulating. Zooplankters are microscopic, crustacean animals found in every lake, but at different densities and with varying size distributions. A sufficient population of large-bodied herbivorous zooplankters, preferably species of *Daphnia*, can filter the entire epilimnion each day during the summer as they graze on algae, bacteria, and bits of organic matter. Although some algae are resistant to grazing, continual strong grazing pressure will tend to depress the overall algal abundance and increase transparency. Excessive nutrients may allow growth by resistant algae to overcome this grazing effect, but for any given level of fertility, the presence of large-bodied grazers will maintain the lowest possible algal biomass and highest possible clarity (Lathrop et al. 1999). Where non-algal turbidity is substantial, such grazing may have no observable effect, but where algae are the primary determinants of clarity, a variety of benefits are possible.

It has also been suggested that algae-eating fish might control algal biomass if stocked in sufficient quantities. As there are no native species of fish in the USA that consume sufficient quantities of algae as their diet, this would involve introduction of a non-native species, probably of tropical origin (e.g., certain species of *Tilapia*). Given the track record of introduced species (Mills et al. 1994), this does not appear to be a desirable approach, and many states have banned such introductions. Additionally, the excreted nutrients from such a fish population might support the growth of as much algae as those fish could consume. Furthermore, no fish can efficiently feed on the smallest algal cells, potentially resulting in a shift toward smaller cell size and greater turbidity per unit of biomass present. Finally, tropical species such as *Tilapia* are unlikely to overwinter in at least the more northern states, limiting the duration of any effect.

Due to the small size of Lake Como and lack of data associated with it's food web, biomanipulation is not an option for Lake Como at this time.

Pathogens

Viral, bacterial and fungal pathogens have each been explored as possible control methods for algae. Ideally, a lake would be inoculated with a pathogen developed to target either a broad spectrum of algal types, or more likely one or a few species of especially obnoxious algae. Such pathogens have been tried experimentally over the years (Lindmark 1979), but none has proven effective and controllable. In dealing with algae, humans may have technological superiority, but we are at an evolutionary disadvantage. The complexity of biological interactions appears beyond our sustained control, and although we can set processes in motion that may produce desired conditions in a lake, those conditions tend to be temporary.

Competition and Allelopathy

Negative interactions between rooted plants and algae might be harnessed to facilitate control of algal biomass, but it is not clear that such a benefit can be derived within a lake without creating a macrophyte nuisance. Plantings for nutrient control generally involve emergent wetland creation within the watershed, although a portion of the lake is sometimes transformed for this purpose. Commercially marketed "nutri-pods" incorporate rooted or floating plants into a floating structure from which excess biomass can be removed as it develops. There has been no scientific documentation of the success of this approach, but it has no apparent adverse ecological impacts as long as native, non-nuisance species are used.

Competition is also employed at the bacterial level, with microbial additives developed largely in the wastewater treatment industry finding application in lakes. According to product literature, these microbes limit the availability of nutrients essential for algal growth, thus reducing the probability of algal blooms. The products surveyed acted primarily as denitrifiers, removing nitrogen from the system and striving to create a nitrogen limitation on growth. However, there is little scientific documentation of effects in lakes, and reduction of nitrogen:phosphorus ratios seems ill-advised in most lakes, as it favors certain types blue-green algae known to create nuisance conditions. This approach has experienced increased publicity just recently, and perhaps some careful studies will be forthcoming.

Plantings for reduced light penetration might also control algae, but there could be many negative side effects of such an effort. Surface-covering growths of duckweed, water hyacinth, or water chestnut could provide such a light barrier, but at great expense to habitat and water quality.

Although senescence of rooted plants often releases nutrients which can support algal blooms, release of allelopathic substances during the more active growth phase of macrophytes may inhibit algal growth. Mat-forming algae found in association with rooted plant beds appear unaffected, but many more planktonic algal species are not abundant when rooted plant growths are dense. Again, this may represent a trade-off between an algal nuisance and a rooted plant nuisance, and many lakes have both.

The use of barley straw appears to have some merit in the control of algal densities (Barrett et al. 1996, Kay 1996, Ridge and Pillinger 1996, Wynn and Langeland 1996), and combines features of algaecides, allelopathy and competition. Preferably added to shallow, moving water or from pond-side digesters, decaying barley straw gives off substances which inhibit algal growth and seem to be particularly effective against blue-green algae. Although this is not a thoroughly understood or widespread technique at this time, research conducted mainly in England has demonstrated that the decomposition of the barley straw produces allelopathic compounds which act as algaecides. Competition for nutrients between heterotrophic decomposers and autotrophic algae appears to favor the heterotrophs after barley straw addition. Stagnant water reduces production of the essential compounds and uptake of nutrients as low oxygen levels in the straw slow decomposition, and highly turbid water also reduces effectiveness.

Doses of barley straw under well oxygenated conditions are typically around 2.5 g/m^2 of pond surface, with doses of 50 g/m^2 or more necessary where initial algal densities are high or flow is limited. Doses of 100 g/m^2 may cause oxygen stress to the pond as decomposition proceeds, but this can be avoided by the use of a land-based digester into which straw is deposited and through which water is pumped as the straw decays.

It is preferable to control algae by limiting available nutrients in comparison to competition and allelopathy for Lake Como.

In-Lake Management Options - Rooted Plant Control

Overabundant rooted and floating vascular plants create a major nuisance for most lake and reservoir users. In extreme cases, particularly in ponds and in shallow, warm, well-lighted lakes and waterways of the southern USA, unwanted plants (defined as weeds) can cover the entire lake surface and fill most of the water column. The presence of plants is necessary to a variety of desirable lake functions, and the individual form of many plants can be viewed as attractive, but weed infestations interfere with recreation, detract from aesthetic value, and can impair habitat as well. They can also introduce significant quantities of nutrients and organic matter to the water column, stimulating algal blooms, causing deleterious dissolved oxygen fluctuations, and providing the precursors of disinfection byproducts.

Macrophytes (vascular plants and visible algal mats) are generally grouped into classes called emergents (represented by alligatorweed and cattails), floating-leaved (water hyacinth and water lilies), and submergents (hydrilla, milfoil and naiads), plus the mats of filamentous algae discussed in the nuisance algae section of this chapter. Understanding the factors that control plant growth is the first step in controlling weeds.

Macrophytes reproduce by producing flowers and seeds and/or by asexual propagation from various fragments and shoots extending from roots. The primary means of reproduction is an extremely important feature of a plant, and will greatly affect the applicability of control methods.

Growth rates of macrophytes, especially non-native species like water hyacinth, hydrilla, and milfoil, can be very high, but is a function of suitable substrate and available light. Submergent plants will grow profusely only where underwater illumination is sufficient. Highly turbid lakes and reservoirs are unlikely to have dense beds of submerged plants. Significant reductions in algal blooms can also enhance light penetration and allow weeds to grow more extensively and densely. High silt loads to a lake can create a favorable plant substrate, but the silt loading may also create severe turbidity that limits growth. Rock, gravel and coarse sand provide limited rooting opportunity, while finer sands, silts and organic mucks can support substantial plant growths. Steep-sided lakes support a much smaller plant community as a consequence of both peripheral substrate and light limitations. A few plants, including water hyacinth, water lettuce, duckweed, and watermeal, can float on the surface with no roots in the sediment, nearly eliminating substrate and light as key control factors.

Most macrophytes obtain most of their nutrition via roots that extend into the sediment. This is an important ecological feature, as they can therefore be abundant in lakes in which nutrient concentrations in the water column have been reduced through watershed management or in-lake measures. When the sediments are either highly organic (very loose mucks) or inorganic (rock to coarse sand), macrophyte growth may be poor because it is more difficult for roots to take hold and to obtain nutrients in these sediment types. In these two extremes, emergent plants may replace submergents in shallow water because their more extensive root systems are better adapted to these conditions.

Plant assemblages and control needs vary with geography. Northern lakes and reservoirs experience weed infestations from non-native and native plant species, but seasonal changes in light and temperature tend to limit nuisance conditions to the summer. As this corresponds with the period of greatest human use of lakes, however, plant management is often desired. However, attitudes on the level of control needed and the forms of control that are acceptable vary greatly from more southern regions.

Setting goals for rooted plant control is a critical planning step and the choice of management technique(s) will be highly dependent upon those goals. A certain amount of plant growth is an ecological necessity in most lakes. Where fishing is the primary objective, substantial littoral bottom coverage is desirable, with some vertical and horizontal structure created by different species of plants to enhance the habitat for different fish species or life stages. For swimming purposes, having no macrophytes seems desirable from a safety perspective, but a low, dense cover in shallow lakes with silty bottoms can minimize turbidity, another safety concern.

Perhaps the simplest axiom for plant management is that if light penetrates to the bottom and the substrate is not rock or cobble, plants will grow. A program intended to eliminate all plants is both unnatural and maintenance intensive, if possible at all. A program to structure the plant community to meet clear goals in an ecologically and ethically sound manner is more appropriate, although potentially still quite expensive.

Table 2 provides an overview of the techniques used to control rooted plants, with notes on the mode of action, advantages, and disadvantages of each technique. Additional details are provided in narrative form below. Additional detail on many techniques can be found in Cooke et al. (1993a) and in Hoyer and Canfield (1997).

Benthic Barriers

The use of benthic barriers, or bottom covers, is predicated upon the principles that rooted plants require light and can not grow through physical barriers. Applications of clay, silt, sand, and gravel have been used for many years, although plants often root in these covers eventually, and current environmental regulations make it difficult to gain approval for such fill deposition. An exception may exist in the reverse layering technique (KVA 1991), in which sand is pumped from underneath a muck or silt layer and deposited as a new layer on top of the muck or silt. This is technically a re-organizing of the sediments, not new filling. Although expensive on a large scale and not applicable where the muck is not underlain by suitable materials, this technique restores the natural lake bottom of some previous time without sediment removal.

Artificial sediment covering materials, including polyethylene, polypropylene, fiberglass, and nylon, have been developed over the last three decades. A variety of solid and porous forms have been used. Manufactured benthic barriers are negatively buoyant materials, usually in sheet form, which can be applied on top of plants to limit light, physically disrupt growth, and allow unfavorable chemical reactions to interfere with further development of plants (Perkins et al. 1980).

In theory, benthic barriers should be a highly effective plant control technique, at least on a localized, area-selective scale. In practice, however, there have been many difficulties in the deployment and maintenance of benthic barriers, limiting their utility in the broad range of field conditions. Benthic barriers can be effectively used in small areas such as dock spaces and swimming beaches to completely terminate plant growth. The creation of access lanes and structural habitat diversity is also practical. Large areas are not often treated, however, because the cost of materials and application is high and maintenance can be problematic (Engel 1984).

TABLE 2. MANAGEMENT OPTIONS FOR CONTROL OF ROOTED AQUATIC PLANTS

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
Physical Controls			
1) Benthic barriers	♦ Mat of variable composition laid on bottom of target area, preventing plant growth ♦ Can cover area for as little as several months or permanently ♦ Maintenance improves effectiveness ♦ Not often intended for use in large areas, usually applied around docks, in boating lanes, and in swimming areas	♦ Highly flexible control ♦ Reduces turbidity from soft bottoms ♦ Can cover undesirable substrate ♦ Can improve fish habitat by creating edge effects	♦ May cause anoxia at sediment-water interface ♦ May limit benthic invertebrates ♦ Non-selective interference with plants in target area ♦ May inhibit spawning/feeding by some fish species
1.a) Porous or loose-weave synthetic materials	♦ Laid on bottom and usually anchored by sparse weights or stakes ♦ Removed and cleaned or flipped and repositioned at least once per year for maximum effectiveness	♦ Allows some escape of gases which may build up underneath ♦ Panels may be flipped in place or removed for relatively easy cleaning or repositioning	♦ Allows some growth through pores ♦ Gas may still build up underneath in some cases, lifting barrier from bottom
1.b) Non-porous or sheet synthetic materials	♦ Laid on bottom and anchored by many stakes, anchors or weights, or by layer of sand ♦ Not typically removed, but may be swept or "blown" clean periodically	♦ Prevents all plant growth until buried by sediment ♦ Minimizes interaction of sediment and water column	♦ Gas build up may cause barrier to float upwards ♦ Strong anchoring makes removal difficult and can hinder maintenance

TABLE 2. MANAGEMENT OPTIONS FOR CONTROL OF ROOTED AQUATIC PLANTS

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
1.c) Sediments of a desirable composition	♦ Sediments may be added on top of existing sediments or plants. ♦ Use of sand or clay can limit plant growths and alter sediment-water interactions. ♦ Sediments can be applied from the surface or suction dredged from below muck layer (reverse layering technique)	♦ Plant biomass can be buried ♦ Seed banks can be buried deeper ♦ Sediment can be made less hospitable to plant growths ♦ Nutrient release from sediments may be reduced ♦ Surface sediment can be made more appealing to human users ♦ Reverse layering requires no addition or removal of sediment	♦ Lake depth may decline ♦ Sediments may sink into or mix with underlying muck ♦ Permitting for added sediment may be difficult ♦ Addition of sediment may cause initial turbidity increase ♦ New sediment may contain nutrients or other contaminants ♦ Generally too expensive for large scale application
2) Dredging	♦ Sediment is physically removed by wet or dry excavation, with deposition in a containment area for dewatering/disposal ♦ Dredging can be applied on a limited basis, but is most often a major restructuring of a severely impacted system ♦ Plants and seed beds are removed and re-growth can be limited by light and/or substrate limitation	♦ Plant removal with some flexibility ♦ Increases water depth ♦ Can reduce pollutant reserves ♦ Can reduce sediment oxygen demand ♦ Can improve spawning habitat for many fish species ♦ Allows complete renovation of aquatic ecosystem	♦ Temporarily removes benthic invertebrates ♦ May create turbidity ♦ May eliminate fish community (complete dry dredging only) ♦ Possible impacts from containment area discharge ♦ Possible impacts from dredged material disposal ♦ Interference with recreation or other uses during dredging ♦ Usually very expensive
2.a) “Dry” excavation	♦ Lake drained or lowered to maximum extent practical ♦ Target material dried to maximum extent possible ♦ Conventional excavation equipment used to remove sediments	♦ Tends to facilitate a very thorough effort ♦ May allow drying of sediments prior to removal ♦ Allows use of less specialized equipment	♦ Eliminates most aquatic biota unless a portion left undrained ♦ Eliminates lake use during dredging

TABLE 2. MANAGEMENT OPTIONS FOR CONTROL OF ROOTED AQUATIC PLANTS

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
2.b) “Wet” excavation	♦ Lake level may be lowered, but sediments not substantially dewatered ♦ Draglines, bucket dredges, or long-reach backhoes used to remove sediment	♦ Requires least preparation time or effort, tends to be least cost dredging approach ♦ May allow use of easily acquired equipment ♦ May preserve most aquatic biota	♦ Usually creates extreme turbidity ♦ Tends to result in sediment deposition in surrounding area ♦ Normally requires intermediate containment area to dry sediments prior to hauling ♦ May cause severe disruption of ecological function ♦ Usually eliminates most lake uses during dredging
2.c) Hydraulic removal	♦ Lake level not reduced ♦ Suction or cutterhead dredges create slurry which is hydraulically pumped to containment area ♦ Slurry is dewatered; sediment retained, water discharged	♦ Creates minimal turbidity and limits impact on biota ♦ Can allow some lake uses during dredging ♦ Allows removal with limited access or shoreline disturbance	♦ Often leaves some sediment behind ♦ Cannot handle extremely coarse or debris-laden materials ♦ Requires sophisticated and more expensive containment area ♦ Requires overflow discharge from containment area
3) Dyes and surface covers	♦ Water-soluble dye is mixed with lake water, thereby limiting light penetration and inhibiting plant growth ♦ Dyes remain in solution until washed out of system. ♦ Opaque sheet material applied to water surface	♦ Light limit on plant growth without high turbidity or great depth ♦ May achieve some control of algae as well ♦ May achieve some selectivity for species tolerant of low light	♦ May not control peripheral or shallow water rooted plants ♦ May cause thermal stratification in shallow ponds ♦ May facilitate anoxia at sediment interface with water ♦ Covers inhibit gas exchange with atmosphere

TABLE 2. MANAGEMENT OPTIONS FOR CONTROL OF ROOTED AQUATIC PLANTS

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
4) Mechanical removal	♦ Plants reduced by mechanical means, possibly with disturbance of soils ♦ Collected plants may be placed on shore for composting or other disposal ♦ Wide range of techniques employed, from manual to highly mechanized ♦ Application once or twice per year usually needed	♦ Highly flexible control ♦ May remove other debris ♦ Can balance habitat and recreational needs	♦ Possible impacts on aquatic fauna ♦ Non-selective removal of plants in treated area ♦ Possible spread of undesirable species by fragmentation ♦ Possible generation of turbidity
4.a) Hand pulling	♦ Plants uprooted by hand ("weeding") and preferably removed	♦ Highly selective technique	♦ Labor intensive
4.b) Cutting (without collection)	♦ Plants cut in place above roots without being harvested	♦ Generally efficient and less expensive than complete harvesting	♦ Leaves root systems and part of plant for re-growth ♦ Leaves cut vegetation to decay or to re-root ♦ Not selective within applied area
4.c) Harvesting (with collection)	♦ Plants cut at depth of 2-10 ft and collected for removal from lake	♦ Allows plant removal on greater scale	♦ Limited depth of operation ♦ Usually leaves fragments which may re-root and spread infestation ♦ May impact lake fauna ♦ Not selective within applied area ♦ More expensive than cutting
4.d) Rototilling	♦ Plants, root systems, and surrounding sediment disturbed with mechanical blades	♦ Can thoroughly disrupt entire plant	♦ Usually leaves fragments which may re-root and spread infestation ♦ May impact lake fauna ♦ Not selective within applied area ♦ Creates substantial turbidity ♦ More expensive than harvesting

TABLE 2. MANAGEMENT OPTIONS FOR CONTROL OF ROOTED AQUATIC PLANTS

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
4.e) Hydroraking	♦ Plants, root systems and surrounding sediment and debris disturbed with mechanical rake, part of material usually collected and removed from lake	♦ Can thoroughly disrupt entire plant ♦ Also allows removal of stumps or other obstructions	♦ Usually leaves fragments which may re-root and spread infestation ♦ May impact lake fauna ♦ Not selective within applied area ♦ Creates substantial turbidity ♦ More expensive than harvesting
5) Water level control	♦ Lowering or raising the water level to create an inhospitable environment for some or all aquatic plants ♦ Disrupts plant life cycle by dessication, freezing, or light limitation	♦ Requires only outlet control to affect large area ♦ Provides widespread control in increments of water depth ♦ Complements certain other techniques (dredging, flushing)	♦ Potential issues with water supply ♦ Potential issues with flooding ♦ Potential impacts to non-target flora and fauna
5.a) Drawdown	♦ Lowering of water over winter period allows desiccation, freezing, and physical disruption of plants, roots and seed beds ♦ Timing and duration of exposure and degree of dewatering are critical aspects ♦ Variable species tolerance to drawdown; emergent species and seed-bearers are less affected ♦ Most effective on annual to once/3 yr. basis	♦ Control with some flexibility ♦ Opportunity for shoreline clean-up/structure repair ♦ Flood control utility ♦ Impacts vegetative propagation species with limited impact to seed producing populations	♦ Possible impacts on contiguous emergent wetlands ♦ Possible effects on overwintering reptiles and amphibians ♦ Possible impairment of well production ♦ Reduction in potential water supply and fire fighting capacity ♦ Alteration of downstream flows ♦ Possible overwinter water level variation ♦ Possible shoreline erosion and slumping ♦ May result in greater nutrient availability for algae

TABLE 2. MANAGEMENT OPTIONS FOR CONTROL OF ROOTED AQUATIC PLANTS

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
5.b) Flooding	♦ Higher water level in the spring can inhibit seed germination and plant growth ♦ Higher flows which are normally associated with elevated water levels can flush seed and plant fragments from system	♦ Where water is available, this can be an inexpensive technique ♦ Plant growth need not be eliminated, merely retarded or delayed ♦ Timing of water level control can selectively favor certain desirable species	♦ Water for raising the level may not be available ♦ Potential peripheral flooding ♦ Possible downstream impacts ♦ Many species may not be affected, and some may be benefitted ♦ Algal nuisances may increase where nutrients are available
Chemical controls 6) Herbicides	♦ Liquid or pelletized herbicides applied to target area or to plants directly ♦ Contact or systemic poisons kill plants or limit growth ♦ Typically requires application every 1-5 yrs	♦ Wide range of control is possible ♦ May be able to selectively eliminate species ♦ May achieve some algae control as well	♦ Possible toxicity to non-target species of plants/animals ♦ Possible downstream impacts; may affect non-target areas within pond ♦ Restrictions of water use for varying time after treatment ♦ Increased oxygen demand from decaying vegetation ♦ Possible recycling of nutrients to allow other growths
6.a) Forms of copper	♦ Contact herbicide ♦ Cellular toxicant, suspected membrane transport disruption ♦ Applied as wide variety of liquid or granular formulations, often in conjunction with polymers or other herbicides	♦ Moderately effective control of some submersed plant species ♦ More often an algal control agent	♦ Toxic to aquatic fauna as a function of concentration, formulation, and ambient water chemistry ♦ Ineffective at colder temperatures ♦ Copper ion persistent; accumulates in sediments or moves downstream

TABLE 2. MANAGEMENT OPTIONS FOR CONTROL OF ROOTED AQUATIC PLANTS

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
6.b) Forms of endothall (7-oxabicyclo [2.2.1] heptane-2,3-dicarboxylic acid)	♦ Contact herbicide with limited translocation potential ♦ Membrane-active chemical which inhibits protein synthesis ♦ Causes structural deterioration ♦ Applied as liquid or granules	♦ Moderate control of some emersed plant species, moderately to highly effective control of floating and submersed species ♦ Limited toxicity to fish at recommended dosages ♦ Rapid action	♦ Non-selective in treated area ♦ Toxic to aquatic fauna (varying degrees by formulation) ♦ Time delays on use for water supply, agriculture and recreation ♦ Safety hazards for applicators
6.c) Forms of diquat (6,7-dihydropyrido [1,2-2',1'-c] pyrazinediium dibromide)	♦ Contact herbicide ♦ Absorbed by foliage but not roots ♦ Strong oxidant; disrupts most cellular functions ♦ Applied as a liquid, sometimes in conjunction with copper	♦ Moderate control of some emersed plant species, moderately to highly effective control of floating or submersed species ♦ Limited toxicity to fish at recommended dosages ♦ Rapid action	♦ Non-selective in treated area ♦ Toxic to zooplankton at recommended dosage ♦ Inactivated by suspended particles; ineffective in muddy waters ♦ Time delays on use for water supply, agriculture and recreation
6.d) Forms of glyphosate (N-[phosphonomethyl glycine)	♦ Contact herbicide ♦ Absorbed through foliage, disrupts enzyme formation and function in uncertain manner ♦ Applied as liquid spray	♦ Moderately to highly effective control of emersed and floating plant species ♦ Can be used selectively, based on application to individual plants ♦ Rapid action ♦ Low toxicity to aquatic fauna at recommended dosages ♦ No time delays for use of treated water	♦ Non-selective in treated area ♦ Inactivation by suspended particles; ineffective in muddy waters ♦ Not for use within 0.5 miles of potable water intakes ♦ Highly corrosive; storage precautions necessary

TABLE 2. MANAGEMENT OPTIONS FOR CONTROL OF ROOTED AQUATIC PLANTS

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
6.e) Forms of 2,4-D (2,4-dichlorophenoxyl acetic acid)	♦ Systemic herbicide ♦ Readily absorbed and translocated throughout plant ♦ Inhibits cell division in new tissue, stimulates growth in older tissue, resulting in gradual cell disruption ♦ Applied as liquid or granules, frequently as part of more complex formulations, preferably during early growth phase of plants	♦ Moderately to highly effective control of a variety of emersed, floating and submersed plants ♦ Can achieve some selectivity through application timing and concentration ♦ Fairly fast action	♦ Variable toxicity to aquatic fauna, depending upon formulation and ambient water chemistry ♦ Time delays for use of treated water for agriculture and recreation ♦ Not for use in water supplies
6.f) Forms of fluridone (1-methyl-3-phenyl-5-[-3-{trifluoromethyl} phenyl]-4[IH]-pyridinone)	♦ Systemic herbicide ♦ Inhibits carotenoid pigment synthesis and impacts photosynthesis ♦ Best applied as liquid or granules during early growth phase of plants	♦ Can be used selectively, based on concentration ♦ Gradual deterioration of affected plants limits impact on oxygen level (BOD) ♦ Effective against several difficult-to-control species ♦ Low toxicity to aquatic fauna	♦ Impacts on non-target plant species possible at higher doses ♦ Extremely soluble and mixable; difficult to perform partial lake treatments ♦ Requires extended contact time
6.g Forms of triclopyr (3,5,6-trichloro-2-pyridinyloxyacetic acid)	♦ Systemic herbicide, registered for experimental aquatic use by cooperators in selected areas only at this time ♦ Readily absorbed by foliage, translocated throughout plant ♦ Disrupts enzyme systems specific to plants ♦ Applied as liquid spray or subsurface injected liquid	♦ Effectively controls many floating and submersed plant species ♦ Can be used selectively, more effective against dicot plant species, including many nuisance species ♦ Effective against several difficult-to-control species ♦ Low toxicity to aquatic fauna ♦ Fast action	♦ Impacts on non-target plant species possible at higher doses ♦ Current time delay of 30 days on consumption of fish from treated areas ♦ Necessary restrictions on use of treated water for supply or recreation not yet certain

TABLE 2. MANAGEMENT OPTIONS FOR CONTROL OF ROOTED AQUATIC PLANTS

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
Biological Controls			
7) Biological introductions	♦ Fish, insects or pathogens which feed on or parasitize plants are added to system to affect control ♦ The most commonly used organism is the grass carp, but the larvae of several insects have been used more recently, and viruses are being tested	♦ Provides potentially continuing control with one treatment ♦ Harnesses biological interactions to produce desired conditions ♦ May produce potentially useful fish biomass as an end product	♦ Typically involves introduction of non-native species ♦ Effects may not be controllable ♦ Plant selectivity may not match desired target species ♦ May adversely affect indigenous species
7.a) Herbivorous fish	♦ Sterile juveniles stocked at density which allows control over multiple years ♦ Growth of individuals offsets losses or may increase herbivorous pressure	♦ May greatly reduce plant biomass in single season ♦ May provide multiple years of control from single stocking ♦ Sterility intended to prevent population perpetuation and allow later adjustments	♦ May eliminate all plant biomass, or impact non-target species more than target forms ♦ Funnels energy into largely unused fish biomass and algae ♦ May drastically alter habitat ♦ May escape to new habitats upstream or downstream ♦ May not always be sterile; population control uncertain
7.b) Herbivorous insects	♦ Larvae or adults stocked at density intended to allow control with limited growth ♦ Intended to selectively control target species ♦ Milfoil weevil is best known, but still experimental	♦ Involves species native to region, or even targeted lake ♦ Expected to have no negative effect on non-target species ♦ May facilitate longer term control with limited management	♦ Population ecology suggests incomplete control likely ♦ Oscillating cycle of control and re-growth likely ♦ Predation by fish may complicate control ♦ Other lake management actions may interfere with success
7.c) Fungal/bacterial/viral pathogens	♦ Inoculum used to seed lake or target plant patch ♦ Growth of pathogen population expected to achieve control over target species	♦ May be highly species specific ♦ May provide substantial control after minimal inoculation effort	♦ Largely experimental; effectiveness and longevity of control not well known ♦ Infection ecology suggests incomplete control likely ♦ Possible side effects not well understood

TABLE 2. MANAGEMENT OPTIONS FOR CONTROL OF ROOTED AQUATIC PLANTS

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
7.d) Selective plantings	♦ Establishment of plant assemblage resistant to undesirable species ♦ Plants introduced as seeds, cuttings or whole plants	♦ Can restore native assemblage ♦ Can encourage assemblage most suitable to lake uses ♦ Supplements targeted species removal techniques	♦ Largely experimental at this time; few well documented cases ♦ Nuisance species may eventually outcompete established assemblage ♦ Introduced species may become nuisances

Benthic barrier problems of prime concern include long-term integrity of the barrier, billowing caused by trapped gases, accumulation of sediment on top of barriers, and growth of plants on porous barriers. Additionally, benthic barriers are non-selective, killing all plants over which they are applied. Oxygen depression and related chemical changes under the barrier result in reductions in the density and diversity of the benthic invertebrate community, but recovery is rapid once the barrier is removed (Ussery et al. 1997). One final problem is the tendency of products to come and go without much stability in the market. Few of the barrier materials on the market at any time continue to be available for more than 5 to 10 years; most need to be made in bulk to keep costs down, yet cost remains high enough to hinder demand and reduce bulk use.

Successful use is related to selection of materials and the quality of the application. As a result of field experience with benthic barriers, several guidelines can be offered:

- Porous barriers will be subject to less billowing, but will allow settling plant fragments to root and growth; annual maintenance is therefore essential
- Solid barriers will generally prevent rooting in the absence of sediment accumulations, but will billow after enough gases accumulate; venting and strong anchoring are essential in most cases
- Plants under the barrier will usually die completely after about a month, with solid barriers more effective than porous ones in killing the whole plant; barriers of sufficient tensile strength can then be moved to a new location, although continued presence of solid barriers restricts recolonization.

Proper application requires that the screens be placed flush with the sediment surface and staked or securely anchored. This may be difficult to accomplish over dense plant growth, and a winter drawdown can provide an ideal opportunity for application. Late spring application has also been effective, however, despite the presence of plant growths at that time, and barriers applied in early May have been removed in mid-June with no substantial plant growth through the summer (Wagner 1991). Scuba divers normally apply the covers in deeper water, which greatly increases labor costs. Bottom barriers will accumulate sediment deposits in most cases, which allows plant fragments to root. Barriers must then be cleaned, necessitating either removal or laborious in-place maintenance.

Despite application and maintenance issues, benthic barriers are a very effective tool. In northern waters, benthic barriers are capable of providing control of milfoil on at least a localized basis (Engel 1984, Perkins et al. 1980, Helsel et al. 1996), and have such desirable side benefits as creating more edge habitat within dense plant assemblages and minimizing turbidity generation from fine bottom sediments.

As an example, benthic barriers have been used at Lake George since 1986 (Madsen et al. 1989, Eichler et al. 1995). Dartek™ was initially installed over 3 acres of milfoil in two areas, and was successful in controlling milfoil within the treated area for about 3 years. No supplementary management actions were conducted, however, and peripheral growths expanded and achieved bed densities in 1989. Sediment accumulation in one area exposed to frequent traffic by large boats was sufficient to allow dense growths of milfoil on portions of the barrier in 1990; those growths were still present in 1995.

Aquascreen™ (a fine mesh material) and Palco Pond Liner™ (an impermeable membrane) were installed at 8 sites in Lake George in 1990. Both barrier types were initially successful in eliminating targeted beds, although recolonization of Aquascreen left in place without annual maintenance was far greater than for the Palco material.

Dartek and Aquascreen are no longer commercially available, but a mesh product very similar to Aquascreen, called Aquatic Weed Net™, is now on the market. Palco is now made by a different supplier, but can be obtained. An additional product, Texel™, is a felt-like sheeting material which has not been tried in Lake George but is potentially applicable and is slightly less costly than the other materials.

Study of recolonization of areas of Lake George where benthic barrier has been removed (Eichler et al. 1995) reveals that both native species and milfoil were found to colonize exposed areas, but that milfoil dominance was not regained for at least two growing seasons. However, milfoil recolonization was not completely prevented in most cases. In Lake George, cover by plants was sparse for at least the first month after removal of the barrier and did not typically exceed 74% after two growing seasons, providing ample opportunity for milfoil invasion.

Recolonization of plants following benthic barrier application and removal in two swimming areas in Great Pond, Massachusetts, has also been studied (Wagner 1991). These applications were for the purpose of improving swimming safety, and did not involve control of any invasive non-native species. In one swimming area, a plant community not differentiable from the original assemblage was restored mainly from seed germination within one to two years after barrier removal. Only one new species was detected, a native plant found in neighboring ponds, and then only as a very minor component of the post-treatment plant community. In the other swimming area, foot traffic in sections which were considered unusable prior to treatment resulted in continued minimal plant growth.

On a localized scale in Lake Como bottom barriers could be an effective means to control rooted plant growths. Except where the substrate is gravelly, plant growths can be expected and may require control to limit interference with recreation. Bottom barriers offer a smaller scale, localized approach to managing plant nuisances and would have minimal consequences on the overall lake ecosystem.

Dredging

Dredging works as a plant control technique when either a light limitation on growth is imposed through increased water depth or when enough “soft” sediment (muck, clay, silt and fine sand) is removed to reveal a less hospitable substrate (typically rock, gravel or coarse sand). The only exception may be suction dredging, whereby a target species can be reduced or possibly eliminated by removing whole plants and any associated seed banks. Suction dredging might more appropriately be considered a form of harvesting, however, as plants are extracted from the bottom by SCUBA divers operating the suction dredge and sediment is often returned to the lake.

The amount of sediment removed, and hence the new depth and associated light penetration, is critical to successful long-term control of rooted, submerged plants. There appears to be a direct relation between water transparency, as determined with a Secchi disk, and the maximum depth of colonization (MDC) by macrophytes. Canfield et al. (1985) provided equations to estimate MDC in Florida and Wisconsin from Secchi disk measurements:

State	Equation
Florida	$\log MDC = 0.42 \log SD + 0.41$
Wisconsin	$\log MDC = 0.79 \log SD + 0.25$

where SD = Secchi depth in meters

It is likely that non-algal turbidity limits light availablity and hence limits some rooted plant growths in Lake Como. It is expected that if turbidity decreases, more rooted aquatic plants will grow. Based on the predicted maximum depth obtained by dredging, plants could colonizes the entire open water habitat of Lake Como. A relatively clear lake often needs a maximum depth of greater than 12 ft to obtain areas where plant could not grow (provided the appropriate substrate was available).

If the soft sediment accumulations which are supporting rooted plant nuisances are not especially thick, it may be possible to create a substrate limitation before a light-limiting depth is reached. If dredging exposes rock ledge or cobble, and all soft sediment can be removed, there will be little rooted plant growth. Yet such circumstances are rare to non-existent; either the sediments grade slowly into coarser materials, or it is virtually impossible to remove all fine sediments from the spaces around the rock or cobble. Consequently, at least 25% regrowth is to be expected when light penetrates to the bottom.

Dredging can be accomplished by multiple methods which can be conveniently grouped into four categories:

- Dry excavation, in which the lake is drained to the extent possible, the sediments are dewatered by gravity and/or pumping, and sediments are removed with conventional excavation equipment such as backhoes, bulldozers, or draglines.
- Wet excavation, in which the lake is not drained or only partially drawn down (to minimize downstream flows), with excavation of wet sediments by various bucket dredges mounted on cranes or amphibious excavators.
- Hydraulic dredging, requiring a substantial amount of water in the lake to float the dredge and provide a transport medium for sediment. Hydraulic dredges are typically equipped with a cutterhead which loosens sediments that are then mixed with water and transported as a

pumped slurry of 80 to 90% water and 10 to 20% solids through a pipeline that traverses the lake from the dredging site to a disposal area.

- Pneumatic dredging, in which air pressure is used to pump sediments out of the lake at a higher solids content (reported as 50 to 70%). This would seem to be a highly desirable approach, given containment area limitation in many cases and more rapid drying with higher solids content. However, few of these dredges are operating within North America, and there is little freshwater experience upon which to base a review. Considerations are much like those for hydraulic dredging, but no further text will be devoted to this technique.

Experience with dredging for rooted plant control has had mixed results. As with dredging for algal control, failures are invariably linked to incomplete pre-dredging assessment and planning. Control through light limitation appears more successful than control through substrate limitation, largely as a function of the difficulty of removing all soft sediment from shallow areas. Dry dredging projects appear to result in more thorough soft sediment removal, mainly because equipment operators can visually observe the results of dredging as it takes place. Hydraulic dredging in areas with dense weed beds can result in frequent clogging of the pipeline to the slurry discharge area, suggesting the need for some form of temporary plant control (most often herbicides or harvesting) prior to hydraulic dredging.

The potential for serious negative impacts by dredging on the lake and surrounding area is very high. Many of these problems are short-lived, however, and can be minimized with proper planning. It should be kept in mind, however, that dredging represents a major re-engineering of a lake, and should not be undertaken without clear recognition of its full impact, positive and negative.

Dredging of Lake Como to control rooted plants would involve creating a substrate limitation in water <12 ft deep. Although expensive, this technique is a viable and recommended option for Lake Como together with watershed nutrient input control.

Light Limitation with Dyes and Surface Covers

The use of dyes as algal control agents was discussed previously, and the same dyes are used in rooted plant control efforts. Dyes are used to limit light penetration and therefore restrict the depth at which rooted plants can grow. They tend to reduce the maximum depth of plant growth, but have little effect in shallow water (<4 ft deep). They are only selective in the sense that they favor species tolerant of low light or with sufficient food reserves to support an extended growth period (during which a stem could reach the lighted zone). In lakes with high transparency but only moderate depth and ample soft sediment accumulations, dyes may provide open water where little would otherwise exist. Repeated treatment will be necessary, as the dye flushes out of the system. Dyes are typically permitted under the same process as herbicides, despite their radically different mode of action.

Surface shading has received little attention as a rooted plant control technique, probably as a function of potential interference with recreational pursuits which are a goal of most rooted plant control programs. Polyethylene sheets, floated on the lake surface, were used by Mayhew and Runkel (1962) to shade weeds. They found that two to three weeks of cover were sufficient to eliminate all species of pondweeds (*Potamogeton* spp.) for the summer if the sheets were applied in spring before plants grew to maturity. Coontail was also controlled, but the generally desirable macroalga *Chara* was not. This procedure should be a useful and inexpensive alternative to traditional methods of weed control in small areas such as docks and beaches, and could be timed to yield results acceptable to summer human users with minimal negative impacts to system ecology.

The artificial color imparted by dyes limits the likely acceptability of this technique at Lake Como, while the potential interference of surface covers with recreation limits their utility. However, surface covers might be used on a localized basis much like bottom barriers. The key would be to have the surface cover in place during spring to retard growths, then remove it for the recreational season.

Mechanical Removal

There are many variations on mechanical removal of macrophytes. Table 2 breaks these varied techniques into hand pulling, cutting without collection, harvesting with collection, rototilling, and hydroraking. Suction dredging, addressed in the dredging section, could also be included here, as it is primarily intended to remove plant biomass. Other classification systems are undoubtedly applicable; this is a diverse collection of methods linked by the commonality of physically attacking the targeted plants. These techniques are often cited as being analogous to mowing the lawn (cutting or harvesting), weeding the garden (hand pulling), or tilling the soil (rototilling or hydroraking), and these are reasonable comparisons. Mechanical management of aquatic plants is not much different from managing terrestrial plants, except for the complications imposed by the water.

Hand pulling is exactly what it sounds like; a snorkeler or diver surveys an area and selectively pulls out unwanted plants on an individual basis. This is a highly selective technique, and a labor intensive one. It is well suited to vigilant efforts to keep out invasive species which have not yet become established in the lake or area of concern. Hand pulling can also effectively address non-dominant growths of undesirable species in mixed assemblages, or small patches of plants targeted for removal. This technique is not suited to large scale efforts, especially when the target species or assemblage occurs in dense or expansive beds.

Hand harvesting records for Eurasian watermilfoil in Lake George in New York for 1989-91 (DFWI 1991) reveal the following:

- First time harvest averaged 90 plants per person-hour
- Second time harvest (re-visit of harvested sites the next year) averaged 41 plants per person-hour

- Except for one site which experienced substantial regrowth, the year after initial harvest regrowth was 20-40% of the initial density
- Regrowth two years after initial harvest averaged <10% of the initial density
- Although plant density and total harvesting effort declines with successive harvesting, effort declines more slowly; harvest time per plant therefore increases with decreasing density, mainly as a function of search time.
- Actual harvesting effort directed at 12 sites was 169 hours for first time harvest and 90 hours for second time harvest

Hand pulling can be augmented by various tools, including a wide assortment of rakes, cutting tools, water jetting devices, nets and other collection devices. McComas (1993) provides an extensive and enjoyable review of options. Use of these tools transitions into the next two categories, macrophyte cutting and harvesting. Suction dredging is also used to augment hand pulling, allowing a higher rate of pulling in a targeted area, as the diver/snorkeler does not have to carry pulled plants to a disposal point.

Cutting is also exactly what it appears to be. A blade of some kind is applied to plants, severing the active apical meristem (location of growth) and possibly much more of the plant from the remaining rooted portion. Regrowth is expected, and in some species that regrowth is so rapid that it negates the benefits of the cutting in only a week or two. If the plant can be cut close enough to the bottom, or repeatedly, it will sometimes die, but this is more the exception than the rule. Cutting is defined here as an operation which does not involve collecting the plants once they are cut, so impacts to dissolved oxygen are possible in large scale cutting operations.

The most high technology cutting technique involves the use of mechanized barges normally associated with harvesting operations, in which plants are normally collected for out-of-lake disposal. In its use as a cutting technology, the “harvester” cuts the plants but does not collect them. A recent modification in this technique employs a grinding apparatus which ensures that viable plant fragments are minimized after processing. There is a distinct potential for dissolved oxygen impacts as the plant biomass decays, much like what would be expected from most herbicide treatments.

Harvesting may involve collection in nets or small boats towed by the person collecting the weeds, or can employ smaller boat-mounted cutting tools which haul the cut biomass into the boat for eventual disposal on land, or can be accomplished with larger, commercial machines with numerous blades, a conveyor system, and a substantial storage area for cut plants. Offloading accessories are available, allowing easy transfer of weeds from the harvester to trucks which haul the weeds to a composting area. Choice of equipment is really a question of scale, with most larger harvesting operations employing commercially manufactured machines built to specifications suited to the job. Some lake associations choose to purchase and operate harvesters, while others prefer to contract harvesting services to a firm which specializes in lake management efforts.

Cutting rates for commercial harvesters tend to range from about 0.2 to 0.6 acres per hour, depending on machine size and operator ability, but the range of possible rates is larger. Even at the highest conceivable rate, harvesting is a slow process which may leave some lake users dissatisfied with progress in controlling aquatic plants. Weed disposal is not usually a problem, in part because lakeshore residents and farmers often will use the weeds as mulch and fertilizer. Also, since aquatic plants are more than 90 percent water, their dry bulk is comparatively small. Key issues in choosing a harvester include depth of operation, volume and weight of plants which can be stored, reliability and ease of maintenance, along with a host of details regarding the hydraulic system and other mechanical design features.

Rototilling and the use of cultivation equipment are newer procedures with a limited track record (Newroth and Soar, 1986). A rototiller is a barge-like machine with a hydraulically operated tillage device that can be lowered to depths of 10 to 12 feet for the purpose of tearing out roots. Also, if the water level in the lake can be drawn down, cultivation equipment pulled behind tractors on firm sediments can achieve 90 percent root removal. Potential impacts to non-target organisms and water quality are substantial, but where severe weed infestations exist, this technique could be appropriate.

Hydroraking involves the equivalent of a floating backhoe, usually outfitted with a York rake which looks like certain farm implements for tilling or moving silage. The tines of the rake attachment are moved through the sediment, ripping out thick root masses and associated sediment and debris. A hydrorake can be a very effective tool for removing submerged stumps, water lily root masses, or floating islands. Use of a hydrorake is not a delicate operation, however, and will create substantial turbidity and plant fragments. Hydroraking in combination with a harvester can remove most forms of vegetation encountered in lakes.

Most mechanical plant removal operations are successful in producing at least temporary relief from nuisance plants and in removing organic matter and nutrients without the addition of a potentially deleterious substance. Plant regrowth can be very rapid (days or weeks), especially in southern waters where midsummer growth rates of water hyacinth can exceed the rate at which they can be harvested. Harvesting may reduce plant diversity in some cases, and resultant open areas are candidates for colonization by invasive species, but most potential problems can be avoided by proper program planning.

A bay of LaDue Reservoir (Geauga County, Ohio) was harvested in July 1982 by the traditional method in which the operator treats the weed bed like a residential lawn and simply mows the area. Stumps of Eurasian watermilfoil plants about 0.5 to 3 inches in height were left, and complete regrowth occurred in 21 days. In contrast, the slower method of lowering the cutter blade about 1 inch into the soft lake mud produced season-long control of milfoil by tearing out roots (Conyers and Cooke, 1983). However, this cutting technique is of little value where sediments are very stiff or in deeper water where the length of the cutter bar can not reach the mud. There is evidence of a carry-over effect (less growth in the subsequent year), especially if an area has had multiple harvests in one season.

Some weed species are more sensitive to harvesting than others. Nicholson (1981) has suggested that harvesting was responsible for spreading milfoil in Chautauqua Lake, New York, because the harvester spread fragments of plants from which new growths could begin. On the other hand, milfoil has become the dominant plant in many northeastern lakes without harvesting programs in less than 5 years after initial appearance (Wagner, pers. obs.). Timely harvesting of species which depend upon seeds for annual re-establishment can eventually limit the extent of those species, but the viability of seeds placed in the sediment over years prior to harvesting can minimize impacts for several years to a decade. Extensive harvest of water chestnut in impounded sections of the Charles River in Boston in 1996 had no observable effect on 1997 growths of that plant. Harvesting was repeated in 1997, and growths in 1998 were much reduced, but it is not clear if it was the effect of harvesting or very high spring water level in 1998 which was responsible (Smith, pers. comm.).

There are few data on the actual restorative effects of harvesting, in the sense of removing significant amounts of nutrients or in reducing the release of nutrients and organic matter to the water column from plant senescence. If nutrient inputs are moderate and weed density is high, as much as 40 to 60% of net annual phosphorus loading could be removed with intense harvesting. This would be a significant nutrient removal in many cases. On the other hand, harvesting itself can increase water column phosphorus concentration either through mechanical disturbance of sediments or by enhancing conditions for phosphorus release from sediments.

Any form of harvesting at Lake Como is likely to be a short-term maintenance effort that would have to be repeated annually. The use of herbicides would be a more cost-effective approach in this case.

Water Level Control

Historically, water level drawdown has been used in waterfowl impoundments and wetlands for periods of a year or more, including the growing season, to improve the quality of wetlands for waterfowl breeding and feeding habitat (Kadlek 1962, Harris and Marshall 1963). It has also been a common fishery management method. Until a few decades ago, drawdowns of recreational lakes were primarily for the purpose of flood control and allowing access for clean ups and repairs to structures, with macrophyte control as an auxiliary benefit. While this technique is not effective on all submergent species, it does decrease the abundance of some of the chief nuisance species, particularly those which rely on vegetative propagules for overwintering and expansion (Cooke et al. 1993a). If there is an existing drawdown capability, lowering the water level provides an inexpensive means to control some macrophytes. Additional benefits may include opportunities for shoreline maintenance and oxidation or removal of nutrient-rich sediments.

The ability to control the water level in a lake is affected by area precipitation pattern, system hydrology, lake morphometry, and the outlet structure. The base elevation of the outlet or associated subsurface pipe(s) will usually set the maximum drawdown level, while the capacity of the outlet to pass water and the pattern of water inflow to the lake will determine if that base elevation can be achieved and maintained. In some cases, sedimentation of an outlet channel or other obstructions may control the maximum drawdown level.

Several factors affect the success of drawdown with respect to plant control. While drying of plants during drawdowns in southern areas may provide some control, the additional impact of freezing is substantial, making drawdown a more effective strategy for northern lakes during late fall and winter. However, a mild winter or one with early and persistent snow may not provide the necessary level of drying and freezing. The presence of high levels of groundwater seepage into the lake may mitigate or negate destructive effects on target submergent species by keeping the area moist and unfrozen. The presence of extensive seed beds may result in rapid re-establishment of previously occurring or new and equally undesirable plant species. Recolonization from nearby areas may be rapid, and the response of macrophyte species to drawdown is quite variable (Table 3).

Drawdown has a long and largely successful history, even if not always intended as a plant control technique (Dunst et al. 1974, Wlosinski and Koljord 1996). Winter drawdowns of Candlewood Lake in Connecticut (Siver et al. 1986) reduced nuisance species by as much as 90% after initial drawdown. Drawdowns in Wisconsin lakes have resulted in reductions in plant coverage and biomass of 40 to 92% in targeted areas (Dunst et al. 1974). In one Wisconsin case, Beard (1973) reported that winter drawdown of Murphy Flowage opened 64 out of 75 acres to recreation and improved fishing.

The effect of drawdown is not always predictable or desirable, however. Reductions in plant biomass of 44 to 57% were observed in Blue Lake in Oregon (Geiger 1983) following drawdown, but certain nuisance species actually increased and herbicides were eventually applied to regain control. Drawdown of Lake Bomoseen in Vermont (VANR 1990) caused a major reduction in many species, many of which were not targeted for biomass reductions. Reviewing drawdown effectiveness in a variety of lakes, Nichols and Shaw (1983) noted the species-specific effects of drawdown, with a number of possible benefits and drawbacks. A system-specific review of likely and potential impacts is highly advisable prior to conducting a drawdown.

Desirable side effects associated with drawdowns include the opportunity to clean up the shoreline, repair previous erosion damage, repair docks and retaining walls, search for septic system breakout, and physically improve fish spawning areas (Nichols and Shaw 1983, Cooke et al. 1993a, WDNR 1989). The attendant concentration of forage fish and game fish in the same areas may be viewed as a benefit of most drawdowns (Cooke et al. 1993a), although not all fishery professionals agree. Since emergent shoreline vegetation tends to be favored by drawdowns, populations of furbearers are expected to benefit (WDNR 1989). The consolidation of loose sediments and sloughing of soft sediment deposits into deeper water is perceived as a benefit in many cases, at least by shoreline homeowners (Cooke et al. 1993a, WDNR 1989).

Undesirable possible side effects of drawdown include loss or reduction of desirable plant species, facilitation of invasion by drawdown-resistant undesirable plants, reduced attractiveness to waterfowl (considered an advantage by some), possible fishkills if oxygen demand exceeds re-aeration during a prolonged drawdown, altered littoral habitat for fish and invertebrates, mortality among hibernating reptiles and amphibians, impacts to connected wetlands, shoreline erosion during drawdown, loss of aesthetic appeal during drawdown, more frequent algal blooms after refill in some cases, reduction in water supply, impairment of recreational access during the drawdown, and downstream flow impacts (Nichols and Shaw 1983, Cooke et al. 1993a). Careful planning can often avoid many of these negative side effects, but managers should be aware of the potential consequences of any management action.

Desirable flood storage capacity will increase during a drawdown, but associated alteration of the downstream flow regime may have some negative impacts. Once the target drawdown level is achieved, there should be little alteration of downstream flow. However, downstream flows must necessarily be greater during the actual drawdown than they would be if no drawdown was conducted. The key to managing downstream impacts is to minimize erosion and keep flows within an acceptable natural range.

TABLE 3
ANTICIPATED RESPONSES OF SOME WETLAND AND AQUATIC PLANTS TO WINTER WATER LEVEL DRAWDOWN

	Change in Relative Abundance		
	Increase	**No Change**	**Decrease**
Acorus calamus (sweet flag)	E		
Alternanthera philoxeroides (alligator weed)	E		
Asclepias incarnata (swamp milkweed)			E
Brasenia schreberi (watershield)			S
Cabomba caroliniana (fanwort)			S
Cephalanthus occidentalis (buttonbush)	E		
Ceratophyllum demersum (coontail)			S
Egeria densa (Brazilian Elodea)			S
Eichhornia crassipes (water hyacinth)		E/S	
Eleocharis acicularis (needle spikerush)	S	S	S
Elodea canadensis (waterweed)	S	S	S
Glyceria borealis (mannagrass)	E		
Hydrilla verticllata (hydrilla)	S		
Leersia oryzoides (rice cutgrass)	E		
Myrica gale (sweetgale)		E	
Myriophyllum spp. (milfoil)			S
Najas flexilis (bushy pondweed)	S		
Najas guadalupensis (southern naiad)			S
Nuphar spp. (yellow water lily)			E/S
*Nymphaea odorata (*water lily)			S
Polygonum amphibium (water smartweed)		E/S	
Polygonum coccineum (smartweed)	E		
Potamogeton epihydrus (leafy pondweed)	S		
Potamogeton robbinsii (Robbins' pondweed)			S
Potentilla palustris (marsh cinquefoil)			E/S
Scirpus americanus (three square rush)	E		
Scirpus cyperinus (wooly grass)	E		
Scirpus validus (great bulrush)	E		
Sium suave (water parsnip)	E		
Typha latifolia (common cattail)	E	E	
Zizania aquatic (wild rice)		E	

E=emergent growth form; S=submergent growth form; E/S=emergent and submergent forms

After Cooke et al., 1993a

Inability to rapidly refill a lake after drawdown is a standard concern in evaluating the efficacy of a drawdown. There must be enough water entering the lake to refill it within an appropriate timeframe while maintaining an acceptable downstream flow. In northern lakes, the best time for refill is in early spring, when flows typically peak as the snowpack melts and rainfall on frozen ground yields the maximum runoff.

Impairment of water supply during a drawdown is a primary concern of groups served by that supply. Processing or cooling water intakes may be exposed, reducing or eliminating intake capacity. The water level in wells with hydraulic connections to the lake will decline, with the potential for reduced yield, altered water quality and pumping difficulties. Drawdowns of Cedar Lake and Forge Pond in Massachusetts resulted in impairment of well water supplies (Wagner, pers. obs.), but there is little mention of impairment of well production in the reviewed literature.

Recolonization by resistant vegetation is sometimes a function of seed beds and sometimes the result of expansion of shoreline vegetation. *Najas* recolonized areas previously overgrown with *Myriophyllum* after the drawdown of Candlewood Lake in Connecticut (Siver et al. 1986), apparently from seeds that had been in those areas prior to milfoil dominance. Cattails and rushes are the most commonly expanding fringe species (Nichols and Shaw 1983, WDNR 1989). Drawdowns to control nuisance submergent vegetation are usually recommended for alternate years to every third year to prevent domination by resistant plant species (Cooke et al. 1993a), although drawdown may be practiced at a higher frequency to gain initial control of target species.

Recreational facilities and pursuits may be adversely impacted during a drawdown. Swimming areas will shrink and beach areas will enlarge during a drawdown. Boating may be restricted both by available lake area and by access to the lake. Again, winter drawdown will avoid most of these disadvantages, although lack of control over winter water levels can make ice conditions unsafe for fishing or skating. Additionally, outlet structures, docks and retaining walls may be subject to damage from freeze/thaw processes during overwinter drawdowns, if the water level is not lowered beyond all contact with structures.

Carefully planned water level fluctuation can be a useful technique to check nuisance macrophytes and periodically rejuvenate wetland diversity. Planned disturbance is always a threshold phenomenon; a little is beneficial, too much leads to overall ecosystem decline. The depth, duration, timing and frequency of the drawdown are therefore critical elements in devising the most beneficial program.

If technically feasible at Lake Como, drawdown could be a valuable, low-cost technique for the control of rooted aquatic plants. However, the lack of a suitable outlet configuration, relative long refilling time and possibility of non-freeze conditions make this technique highly unpredicable and therefore is not recommended at this time.

Herbicides

Killing nuisance aquatic weeds with chemicals is perhaps the oldest method used to attempt their management. Other than perhaps drawdown, few alternatives to herbicides were widely practiced until relatively recently. There are few aspects of plant control which breed more controversy than chemical control through the use of herbicides, which are a subset of all chemicals known as pesticides. Part of the problem stems from pesticides which have come on the market, enjoyed widespread use, been linked to environmental or human health problems, and been banned from further use. Some left longer term environmental contamination and toxicity problems behind. Many pesticides in use even 20 years ago are not commonly used or even approved for use today. The legacy of books such as Silent Spring (Carson 1962) and Our Stolen Future (Colburn et al. 1997) have done much to raise both public consciousness and wariness of chemicals in the environment.

Yet as chemicals are an integral part of life and the environment, it is logical to seek chemical solutions to such problems as infestations of non-native species which grow to nuisance proportions, just as we seek physical and biological solutions. Current pesticide registration procedures are far more rigorous than in the past. While no pesticide is considered unequivocally "safe", a premise of federal pesticide regulation is that the potential benefits derived from use outweigh the risks when the chemical is used according to label restrictions.

There are only six active ingredients currently approved for use in aquatic herbicides in the USA today, with one additional ingredient in the experimental use phase of the approval process. Westerdahl and Getsinger (1988a, 1988b) provide a detailed discussion of herbicides and related plant susceptibilities.

Copper products have been discussed in some detail in the algal control section. Copper is not typically preferred as a primary herbicide for rooted aquatic plants, but is sometimes part of a broad spectrum formulation intended to reduce the biomass of an entire plant assemblage, especially if it includes a substantial algal component. Copper concentrations should not exceed 1 mg/L in the treated waters.

Endothall is a contact herbicide, attacking plants at the immediate point of contact. Only portions of the plant with which the herbicide can come into contact are killed. It is sold in several formulations: liquid (Aquathol K), granular dipotassium salt (Aquathol), and the di (N, N-dimethyl-alkylanine) salt (Hydrothol) in liquid and granular forms. Effectiveness can range from weeks to months. Most endothall compounds break down readily and are not persistent in the aquatic environment, but the potassium salt forms have been shown to persist in the water for 2 to 46 days.

Endothall acts quickly on susceptible plants, but does not kill roots with which it can not come into contact, and recovery of many plants is rapid. Rapid death of susceptible plants can cause oxygen depletion if decomposition exceeds re-aeration in the treated area, although this can be mitigated by conducting successive partial treatments. Toxicity to invertebrates, fish or humans is not expected to be a problem at the recommended dose, yet water use restrictions are mandated on the label and it is not used in drinking water supplies. Depending upon the formulation, concentrations in treated waters should be limited to 1 to 5 mg/L.

Diquat, like endothall, it is a fast acting contact herbicide, producing results within 2 weeks of application. It is not an especially selective herbicide, and can be toxic to invertebrates, fish, mammals, birds and humans. A domestic water use restriction is normally applied, and this herbicide is not used in drinking water supplies. Regrowth of some species has been rapid (often within the same year) after treatment with diquat in many cases. Concentrations in treated water should not exceed 2 mg/L.

Glyphosate is another contact herbicide. Its aquatic formulation is effective against most emergent or floating-leaved plant species, but not against most submergent species. Its mode of action is not certain, but it appears to disrupt synthesis of necessary compounds within the cell. Rainfall shortly after treatment can negate its effectiveness, and it readily adsorbs to particulates in the water column or to sediments and is inactivated. It is relatively non-toxic to aquatic fauna at recommended doses, and degrades readily into non-toxic components in the aquatic environment. There is no maximum concentration for treated water, but a dose of 0.2 mg/L is recommended.

2,4-D, which is the active ingredient in a variety of commercial herbicide products, has been in use for over 30 years despite claims of undesirable environmental side effects and potential human health effects. This is a systemic herbicide; it is absorbed by roots, leaves and shoots and disrupts cell division throughout the plant. Vegetative propagules such as winter buds, if not connected to the circulatory system of the plant at the time of treatment, are generally unaffected and can grow into new plants. It is therefore important to treat plants early in the season, after growth has become active but before such propagules form.

2,4-D is sold in liquid or granular forms as sodium and potassium salts, as ammonia or amine salts, and as an ester. Doses of 50 to 150 pounds per acre are usual for submersed weeds, most often of the dimethylamine salt or the butoxyethanolester (BEE). This herbicide is particularly effective against Eurasian watermilfoil (granular BEE applied to roots early in the season) and as a foliage spray against water hyacinth. 2,4-D has a short persistence in the water but can be detected in the mud for months.

Experience with granular 2,4-D in the control of nuisance macrophytes has been generally positive, with careful dosage management providing control of such non-native nuisance species as Eurasian watermilfoil with only sublethal damage to many native species (Miller and Trout 1985, Helsel et al. 1996). Recovery of the native community from seed has also been successful. 2,4-D has variable toxicity to fish, depending upon formulation and fish species. The 2,4-D label does not permit use of this herbicide in water used for drinking or other domestic purposes, or for irrigation or watering of livestock. Concentrations in treated water should not exceed 0.1 mg/L.

Recent experiments with plastic curtains to contain waters treated with 2,4-D revealed a loss of only 2-6% of the herbicide to areas outside the target area (Helsel et al. 1996). This approach may mark the beginning of a new wave of more areally selective treatments and integrated rooted plant management.

Fluridone is a systemic herbicide introduced in 1979 (Arnold 1979) and in widespread use since the mid-1980's, although some states have been slow to approve its use. Fluridone currently comes in two formulations, an aqueous suspension and a slow release pellet, although an even slower release pellet is in the development stage. This chemical inhibits carotene synthesis, which in turn exposes the chlorophyll to photodegradation. Most plants are negatively sensitive to sunlight in the absence of protective carotenes, resulting in chlorosis of tissue and death of the entire plant with prolonged exposure to a sufficient concentration of fluridone. Some plants, including Eurasian watermilfoil, are more sensitive to fluridone than others, allowing selective control at low dosages.

For susceptible plants, lethal effects are expressed slowly in response to treatment with fluridone. Existing carotenes must degrade and chlorosis must set in before plants die off; this takes several weeks to several months, with 30-90 days given as the observed range of time for die off to occur after treatment. Fluridone concentrations should be maintained in the lethal range for the target species for at least three weeks, and preferably for six weeks. This presents some difficulty for treatment in areas of substantial water exchange, but the slow rate of die off minimizes the risk of oxygen depletion.

Fluridone is considered to have low toxicity to invertebrates, fish, other aquatic wildlife, and humans. It is not known to be a carcinogen, oncogen, mutagen or teratogen. Research on its degradation products initially suggested some possible effects, but further testing indicated no significant threat. Substantial bioaccumulation has been noted in certain plant species, but not to any great extent in animals. The USEPA has designated a tolerance level of 0.5 ppm (mg/L or mg/kg) for fluridone residues or those of its degradation products in fish or crayfish. The USEPA has set a tolerance limit of 0.15 ppm for fluridone or its degradation products in potable water supplies, although state restrictions are sometimes lower. Control of Eurasian watermilfoil has been achieved for at least a year without significant impact on non-target species at doses <0.01 mg/L (Netherland et al. 1997, Smith and Pullman 1997).

If 40 days of contact time can be achieved, the use of the liquid formulation of fluridone in a single treatment has been very effective. Where dilution is potentially significant, the slow release pellet form of fluridone has generally been the formulation of choice. Gradual release of fluridone, which is 5% of pellet content, can yield a relatively stable concentration. However, pellets have been less effective in areas with highly organic, loose sediments than over sandy or otherwise firm substrates (Haller pers. comm.). A phenomenon termed “plugging” has been observed, resulting in a failure of the active ingredient to be released from the pellet. While some success in soft sediment areas has been achieved (ACT 1994), pellets are likely to be less efficient than multiple, sequential treatments with the liquid formulation in areas with extremely soft sediments and significant flushing.

The active herbicidal ingredient triclopyr is currently experimental for aquatic habitats. It is highly selective and effective against Eurasian watermilfoil at a dose of 1 to 2.5 mg/L. Experimental treatments of aquatic environments (Netherland and Getsinger 1993) have revealed little or no effect on most monocotyledonous naiads and pondweeds, which are mostly valued native species. Its mode of action is to prevent synthesis of plant-specific enzymes, resulting in disruption of growth processes. This herbicide is most effective when applied during the active growth phase of young plants.

Triclopyr is not known to be a carcinogen, oncogen, mutagen or teratogen, and all lethal effects on tested animal populations have occurred at concentrations over 100 times the recommended dosage rate. The experimental label calls for concentrations in potable water of no more than 0.5 mg/L, suggesting that care must be taken to allow sufficient dilution between the point of application and any potable water intakes.

A herbicide treatment can be an effective short-term management procedure to produce a rapid reduction in vegetation for typical periods of weeks to months. In some cases involving fluridone, as many as five years of control can be gained. The use of herbicides to get a major plant nuisance under control is a valid element of long-term management when other means of keeping plant growths under control are then applied. Failure to apply alternative techniques on a smaller scale once the nuisance has been abated places further herbicide treatments in the cosmetic maintenance category; such techniques tend to have poor cost-benefit ratios over the long-term.

Lake managers who choose herbicidal chemicals need to exercise all proper precautions. As shown in Table 4, effectiveness of a given herbicide varies by plant species and therefore the nuisance plants must be carefully identified. Users should follow the herbicide label directions exactly, use only a herbicide registered by EPA for aquatic use, wear protective gear during application, and protect desirable plants. Most states require applicators to be licensed and to have adequate insurance.

TABLE 4. SUSCEPTIBILITY OF COMMON AQUATIC PLANT SPECIES TO HERBICIDES

	Controlled by Herbicide Application				
	Diquat	**Endothal**	**2,4-D**	**Glyphosate**	**Fluridone**
Emergent Species					
Alternanthera philoxeroides (alligator weed)				Y	YY
Dianthera americana (water willow)			Y		
Glyceria borealis (mannagrass)	Y	N	N		
Phragmites spp. (reed grass)				Y	
Sagittaria spp. (arrowhead)	N	N	Y		Y
Scirpus spp. (bulrush)	N	N	Y	Y	Y
Typha spp. (cattail)	Y	N	Y	Y	Y
Floating Species					
Brasenia schreberi (watershield)	N	Y	Y		N
Eichhornia crassipes (water hyacinth)	Y		Y		N
Lemna spp. (duckweed)	Y	N	Y		Y
Nelumbo lutea (American lotus)	N	N	Y	N	
Nuphar spp. (yellow water lily)	N	Y	Y	Y	Y
Nymphaea spp. (white water lily)	N	Y	Y	Y	Y
Wolfia spp. (watermeal)	Y	N	Y		Y
Submerged Species					
Ceratophyllum demersum (coontail)	Y	Y	Y		Y
Cabomba caroliniana (fanwort)	N	N	N	N	Y
Chara spp. (stonewort)	N	N	N	N	
Elodea canadensis (waterweed)	Y		N		Y
Hydrilla verticllata (hydrilla)	Y	Y			Y
Myriophyllum spicatum (milfoil)	Y	Y	Y	N	Y
Najas flexilis (bushy pondweed)	Y	Y	N	N	Y
Najas guadalupensis (southern naiad)	Y	Y	N		Y
Potamogeton amplifolius (largeleaf pondweed)			Y	N	Y
Potamogeton crispus (curlyleaf pondweed)	Y	Y	N		Y
Potamogeton diversifolius (waterthread)	N	Y	N		
Potamogeton natans (floating leaf pondweed)		Y	Y	Y	Y
Potamogeton pectinatus (sago pondweed)	Y	Y	N		Y
Potamogeton illinoensis (Illinois pondweed)					Y
Ranunculus spp. (buttercup)	Y		Y		

Adapted from Nichols 1986. Y=Yes, N=No, blank=uncertain

Note: Chara spp. (stonewort) can be controlled with copper, which also enhances the performance of Diquat on Eichhornia crassipes (water hyacinth).

Important questions to be answered before adopting a management program involving herbicides include:

- What is the acreage and volume of the area(s) to be treated? Proper dosage is based upon these facts.
- What plant species are to be controlled? This will determine the herbicide and dose to be used.
- What will the long-term costs of this decision be? Most herbicides must be reapplied annually, in some cases two to three times per growing season.
- How is this waterbody used? Many herbicides have restrictions of a day to two weeks on water use following application.
- Is the applicator licensed and insured, and has a permit been obtained from the appropriate regulatory agency? All are necessary prior to treatment.

Shireman et al. (1982) caution that the following lake characteristics almost invariably produce undesirable water quality changes after treatment with a herbicide for weed control:

- High water temperature
- High plant biomass to be controlled
- Shallow, nutrient-rich water
- High percentage of lake area treated
- Closed or non-flowing system

Competent applicators will be cautious in treating a lake with these conditions.

For Lake Como, the potential to control water lilies with fluridone (Sonar) is attractive. Repeated applications of Sonar will be just as expensive as other techniques over a person's lifetime. Longer-term benefits would be realized with dredging and possible treatments after dredging should macrophytes revert to nuisance levels.

Biological Introductions

Significant improvement in our future ability to achieve lasting control of nuisance aquatic vegetation may come from plant-eating or plant-pathogenic biocontrol organisms, or from a combination of current procedures such as harvesting, drawdown, and herbicides with these organisms. Biological control has the objective of achieving control of plants without introducing toxic chemicals or using machinery. It suffers from one ecological drawback; in predator-prey (or parasite-host) relationships, it is rare for the predator to completely eliminate the prey. Consequently, population cycles or oscillations are typically induced for both predator and prey. It is not clear that the magnitude of the upside oscillations in plant populations will be acceptable to human users, and it seems likely that a combination of other techniques with biocontrols may be necessary to achieve lasting, predictable results.

Biological controls include herbivorous fish such as *Ctenopharyngidon idella* (the grass carp), insects such as the aquatic weevil (*Euhrychiopsis lecontei*), and experimental fungal pathogens.

Aside from consumptive approaches (grazing, parasitism), it is also possible to exert competitive pressures, limiting invasive species by maintaining a healthy native assemblage.

The grass carp is a non-native fish (imported around 1962) known to be a voracious consumer of many forms of macrophytes. It has a very high growth rate (about 6 pounds per year at the maximum rate; Smith and Shireman, 1983). This combination of broad diet and high growth rate can produce control or even eradication of plants within several seasons. However, grass carp do not consume aquatic plant species without preference. Generally, they avoid alligatorweed, water hyacinth, cattails, spatterdock, and water lily. These fish prefer plant species such as elodea, pondweeds and hydrilla. Low stocking densities can produce selective grazing on the preferred plant species while other less preferred species, including milfoil, may even increase. Overstocking, on the other hand, may eliminate all plants, contrary to the ecological axiom of oscillating population cycles described previously. Feeding preferences are listed in Nall and Schardt (1980), Van Dyke et al. (1984), and Cooke and Kennedy (1989).

Grass carp are not approved for introduction in all states and is therefore not recommended for Lake Como.

The use of insects to control rooted plants has historically centered on introduced, non-native species. Ten insect species have been imported to the United States under quarantine and have received U.S. Department of Agriculture approval for release to U.S. waters. These insects are confined to the waters of southern states, specifically to control alligatorweed, hydrilla, water lettuce and water hyacinth, and include aquatic larvae of moths, beetles and thrips (Cooke et al. 1993a). These 10 species have life histories that are specific to the host plants and are therefore confined in their distribution to infested areas. They also appear climate-limited to southern states, with the northern range being Georgia and North Carolina. Their reproductive rates are slower than their target plants. Therefore, control is slow, although it can be enhanced by integrated techniques whereby plant densities are reduced at a site with harvesting or herbicides, and insects are concentrated on the remaining plants.

Despite some successes, the track record for biological problem-solving through introduced, non-native species is poor (as many problems seem to have been created as solved), and governmental agencies tend to prefer alternative controls unless there is no practical choice. However, the use of native species in a biomanipulative approach is usually acceptable. Combining biological, chemical and mechanical controls is the basis of integrated pest control, and takes advantage of as many avenues of control as possible for maximum effectiveness. The development of native insects as aquatic plant controls is still in its infancy, but several promising developments have occurred in the last decade, mainly in northern states. The use of larvae of midgeflies, caddisflies, beetles and moths have been explored with some promise (Cooke et al. 1993a). However, the activities of the aquatic weevil *Euhrychiopsis lecontei* have received the most attention in recent years.

Euhrychiopsis lecontei is a native North American species believed to have been associated with northern watermilfoil (*Myriophyllum sibericum*), a species largely replaced by non-native, Eurasian watermilfoil (*M. spicatum*) since the 1940's. The weevil is able to switch plant hosts within the milfoil genus, although to varying degrees and at varying rates depending upon genetic stock and host history (Solarz and Newman 1996). It does not utilize non-milfoil species. Its impact on Eurasian watermilfoil was been documented (Creed and Sheldon 1995, Sheldon and Creed 1995, Sheldon and O'Bryan 1996a) through five years of experimentation under USEPA sponsorship. In controlled trials, the weevil clearly has the ability to impact milfoil plants through structural damage to apical meristems (growth points) and basal stems (plant support). Adults and larvae feed on milfoil, eggs are laid on it, and pupation occurs in burrows in the stem.

Field observations link the weevil to natural milfoil declines in nine Vermont lakes. Additional evidence of weevil-induced crashes without introduction or population augmentation exists for lakes outside Vermont (Creed 1998). Lakewide crashes have generally not been observed in cases where the weevil has been introduced into only part of the lake, although localized damage has been substantial and such widespread control may require more time than current research and monitoring has allowed. As with experience with introduced insect species in the south, the population growth rate of the weevil is usually slower than that of its host plant, necessitating supplemental stocking of weevils for more immediate results. Just what allows the weevil to overtake the milfoil population in the cases where natural control has been observed is still unknown.

Densities of 1-3 weevils per stem appear to collapse milfoil plants, and raising the necessary weevils is a major operation. The State of Vermont devoted considerable resources to rearing weevils for introduction over a two-year period, using them all for just a few targeted sites (Hanson et al. 1995). Weevils are now marketed commercially as a milfoil control, with a recommended stocking rate of 3000 adults per acre. Release is often from cages or onto individual stems; early research involved attaching a stem fragment with a weevil from the lab onto a milfoil plant in the target lake, which was highly labor-intensive.

Although weevils may be amenable to use within an integrated milfoil management approach, interference from competing control techniques has been suggested as a cause for sub-optimal control by weevils (Sheldon and O'Bryan 1996b). Harvesting may directly remove weevils and reduce their density during the growing season. Also, adults are believed to overwinter in debris along the edge of the lake, and techniques such as drawdown, bottom barriers, or sediment removal could negatively impact the weevil population. Extension of lawns to the edge of the water and application of insecticides also represent threats to these milfoil control agents.

Plant pathogens remain largely experimental, despite a long history of interest from researchers. Properties of plant pathogens which make them attractive (Freeman 1977) include:

- High abundance and diversity
- High host specificity
- Non-pathogenicity to non-target organisms
- Ease of dissemination and self-maintenance
- Ability to limit host population without elimination

Fungi are the most common plant pathogens investigated, and control of water hyacinth, hydrilla or Eurasian watermilfoil by this method has been extensively evaluated (Charudattan et al. 1989, Theriot 1989, Gunner et al. 1990, Joye 1990). Results have not been consistent or predictable in most cases, and problems with isolating effective pathogens, overcoming evolutionary advantages of host plants, and delivering sufficient inoculum have limited the utility of this approach to date. However, combination of fungal pathogens and herbicides has shown some recent promise as an integrated technique (Nelson et al. 1998).

Due to the small size of Lake Como and lack of data associated with it’s food web, biomanipulation is not an option for Lake Como at this time.