Myriophyllum spicatum INVASION IN CAVE RUN LAKE, KENTUCKY

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ABSTRACT

Myriophyllum spicatum (Eurasian watermilfoil) (milfoil) is an invasive species that has gone through boom and bust cycles in numerous lakes. Since many mountain reservoirs lack sufficient littoral habitat to support fisheries, the establishment of dense beds of this macrophyte is of interest to local fisherman. Cave Run Lake is an 8,270-acre multipurpose impoundment constructed in 1974 on the Licking River in northeastern Kentucky with a maximum depth of 27 m and a mean depth of 8 m. milfoil began to appear in shallow embayments a few years ago and has radiated extensively. We examined where blooms were occurring to determine if there was any impact of milfoil on water quality in the lake. Infrared aerial photos of the lake were taken during the 1999 and 2000 growing seasons to determine the extent of milfoil propagation. Sediment and water samples were taken in November of 1999 and during July and August 2000 both in and out of milfoil beds. Plant
samples were taken in July 2000. Soil analysis included determination of organic
carbon, total N, and total P content. Water quality parameters measured included:
photosynthetically active radiation (PAR), temperature, oxygen, conductivity, nitrate,
ammonium, and soluble reactive phosphorus (SRP). Plant samples were examined for
dry mass. Comparisons were made between samples inside weed beds to sites with no
weed growth (Student’s t-test, p < 0.05). Only sites with similar depths were
compared. We found no significant difference between soil total N, total P and
organic carbon between weeded and non-weeded zones. We also found no effect of
milfoil on water quality parameters, except PAR. Photosynthetically active radiation
was significantly greater (P < 0.05) in non-weeded zones. Macrophyte dry mass was
highest at intermediate depths between 1.5 m and 4 m. The impact of milfoil on Cave
Run Lake fisheries is unclear. While milfoil provides excellent fishing cover, and has
been shown to aid in angling success, its effects on fish reproduction and predation
are unknown. Long-term effects include potential damage to native plant species due
to its ability to outcompete, and impedance of surface water use because of dense
growth.

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# TABLE OF CONTENTS

<table>
<thead>
<tr>
<th>Section</th>
<th>Page Number</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acceptance Page</td>
<td>ii</td>
</tr>
<tr>
<td>Abstract</td>
<td>iii</td>
</tr>
<tr>
<td>Acknowledgments</td>
<td>v</td>
</tr>
<tr>
<td>Table of Contents</td>
<td>vi</td>
</tr>
<tr>
<td>List of Figures</td>
<td>vii</td>
</tr>
<tr>
<td>1. Introduction</td>
<td>1</td>
</tr>
<tr>
<td>2. Literature Review</td>
<td>4</td>
</tr>
<tr>
<td>2.1 Aquatic Macrophytes in Lakes</td>
<td>4</td>
</tr>
<tr>
<td>2.1.1 Aquatic Plant Establishment</td>
<td>4</td>
</tr>
<tr>
<td>2.1.2 Interactions of Aquatic Macrophytes with other Littoral Biota</td>
<td>7</td>
</tr>
<tr>
<td>2.1.3 Introducing Littoral Flora to an Ecosystem</td>
<td>10</td>
</tr>
<tr>
<td>2.1.4 Undesirable Aquatic Plants</td>
<td>11</td>
</tr>
<tr>
<td>2.1.5 Eurasian Watermilfoil</td>
<td>14</td>
</tr>
<tr>
<td>2.1.6 Role of Aquatic Vegetation in Lake Ecosystem</td>
<td>21</td>
</tr>
<tr>
<td>2.2 Importance of Littoral Zones to Fish Communities</td>
<td>22</td>
</tr>
<tr>
<td>2.3 Muskellunge in Reservoirs</td>
<td>24</td>
</tr>
<tr>
<td>3. Materials and Methods</td>
<td>29</td>
</tr>
<tr>
<td>3.1 Study Site</td>
<td>29</td>
</tr>
<tr>
<td>3.2 Aerial Photography</td>
<td>31</td>
</tr>
<tr>
<td>3.3 Water Quality Measurements</td>
<td>33</td>
</tr>
<tr>
<td>3.4 Substrate Analysis</td>
<td>35</td>
</tr>
<tr>
<td>3.4.1 Sample Collection</td>
<td>35</td>
</tr>
<tr>
<td>3.4.2 Nutrient Analysis</td>
<td>36</td>
</tr>
<tr>
<td>3.4.3 Benthic Insect Analysis</td>
<td>36</td>
</tr>
<tr>
<td>3.5 Aquatic Plant Collection and Analysis</td>
<td>36</td>
</tr>
<tr>
<td>3.6 Data Analysis</td>
<td>37</td>
</tr>
<tr>
<td>4. Results</td>
<td>38</td>
</tr>
<tr>
<td>5. Discussion</td>
<td>45</td>
</tr>
<tr>
<td>5.1 Aerial Photography</td>
<td>45</td>
</tr>
<tr>
<td>5.2 Plants</td>
<td>46</td>
</tr>
<tr>
<td>5.3 Water Quality</td>
<td>47</td>
</tr>
<tr>
<td>5.4 Sediment Composition</td>
<td>48</td>
</tr>
<tr>
<td>5.5 Insects</td>
<td>49</td>
</tr>
<tr>
<td>5.6 Conclusions</td>
<td>49</td>
</tr>
<tr>
<td>6. References Cited</td>
<td>62</td>
</tr>
</tbody>
</table>
# LIST OF FIGURES

<table>
<thead>
<tr>
<th>FIGURE</th>
<th>DESCRIPTION</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.</td>
<td>Myriophyllum spicatum L. Eurasian watermilfoil</td>
</tr>
<tr>
<td>2.</td>
<td>Esox masquinongy Mitchill</td>
</tr>
<tr>
<td>3.</td>
<td>Large Map: Cave Run Lake, Kentucky</td>
</tr>
<tr>
<td>4.</td>
<td>Map of Cave Run Lake with icons demonstrating sampling sites for the present survey</td>
</tr>
<tr>
<td>5.</td>
<td>Vegetation density map of Cave Run Lake, Kentucky; September 2000</td>
</tr>
<tr>
<td>6a.</td>
<td>Mean plant biomass vs. depth at site</td>
</tr>
<tr>
<td>6b.</td>
<td>Mean plant biomass vs. depth (all samples)</td>
</tr>
<tr>
<td>7a.</td>
<td>Average photosynthetically active radiation (PAR); weeded versus non-weeded zones</td>
</tr>
<tr>
<td>7b.</td>
<td>Average water conductivity; weeded versus non-weeded zones</td>
</tr>
<tr>
<td>8a.</td>
<td>Average soluble nitrogen trends; weeded versus non weeded zones</td>
</tr>
<tr>
<td>8b.</td>
<td>Average soluble reactive phosphorus concentration trends; weeded versus non- weeded zones</td>
</tr>
<tr>
<td>8c.</td>
<td>Average pH trends; weeded versus non-weeded zones</td>
</tr>
<tr>
<td>9a.</td>
<td>Average soil organic matter composition trends; weeded versus non-weeded zones</td>
</tr>
<tr>
<td>9b.</td>
<td>Average total soil nitrogen and phosphorus composition trends; weeded versus non-weeded zones</td>
</tr>
<tr>
<td>10.</td>
<td>Number of muskellunge caught on Cave Run Lake, Kentucky; 1991-2000</td>
</tr>
</tbody>
</table>
1. Introduction

Aquatic vegetation affects many ecosystem processes while providing structural habitat for fauna in freshwater lakes. Aquatic macrophytes also provide shelter and food for invertebrates, fishes, waterfowl, and some semi-aquatic mammals. (Carpenter and Lodge 1986; Forsyth et al. 1997; Keast 1984; Randall et al. 1996; Wilcox and Meeker 1992). If aquatic flora increase in abundance and diversity, so will associated fauna (Forsyth et al. 1997; Rosine 1955; Wilcox and Meeker 1992).

Freshwater fish are a multi-billion dollar a year industry in the United States (Downing and Plante 1993). Fish production is significantly higher in lakes with abundant submersed macrophytes (Carpenter and Lodge 1986; Cyr and Peters 1996; Downing and Plante 1992; Rosine 1955; Hinch and Collins 1992; Keast 1984; Randall et al. 1996; Wilcox and Meeker 1992). Submersed macrophytes in lake littoral zones are important habitat for fishes as they provide reproductive habitat, cover for protection, and can harbor rich invertebrate communities (Carpenter and Lodge 1986; Cyr and Peters 1996; Downing and Plante 1992; Rosine 1955; Hinch and Collins 1992; Keast 1984; Randall et al. 1996; Wilcox and Meeker 1992).

Much of the recreational game fishing in the Southeastern U.S. occurs in reservoirs. Littoral vegetation in mountain reservoirs is restricted because the lakes do not collect necessary sediment (Peltier and Welch 1969) and dramatic lake level fluctuations (necessary for flood control) preclude germination, establishment, and
growth (Grace and Wetzel 1978; McDonald 1953; Peltier and Welch 1969; Wetzel 1982).

Muskellunge, *Esox masquinongy*, is a sportfish increasing in popularity in eastern Kentucky reservoirs. Lack of sufficient spawning habitat (shallow waters with vegetative substrate for egg support and relatively low turbidity) makes maintaining the populations in reservoirs a challenge for management personnel (Axon 1981; Dombeck et al. 1984; Johnson and Margenau 1993). Muskellunge exhibit no parental care and egg mortality is high (Dombeck et al. 1984) therefore, muskellunge are maintained by stocking (Axon 1981; Johnson and Margenau 1993). Survival of stocked muskellunge can be low, and hatchery-reared muskellunge are expensive (US$ 2.83-7.50 each for 200mm and larger fingerlings in Wisconsin; Dombeck et al. 1984; Margenau 1992; ) therefore, it would be advantageous to determine ways to increase survival (Johnson and Margenau 1993).

Young-of-year muskellunge require substantial forage fish populations to avoid early mortality. Forage fish populations are often dependent on littoral insect populations for a food source. As reservoirs are simple systems, they often lack the resources necessary to drive this trophic cascade. Consequently, hatcheries must stock forage fish to maintain muskellunge populations (Axon 1981; Johnson and Margenau 1993).

Cave Run Lake, in northeastern Kentucky historically has had a low littoral development because of water level fluctuations for flood control. Recently this has changed because of the introduction of Eurasian watermilfoil (*Myriophyllum*...
spicatum) (milfoil). Normally considered a nuisance weed, and regulated in many
northern states and providences, it has provided the first substantial littoral habitat in
Cave Run Lake.

The objectives of this research are:

1. Map the extent of littoral vegetation (mostly milfoil) within Cave Run
   Lake, Kentucky.
2. Determine if a relationship exists between abiotic factors (soil nutrients,
   depth) and plant establishment.
3. Assess if new vegetative zones are providing habitat for benthic aquatic
   insects.
4. Determine if water quality is affected by vegetation.
2. Literature Review

2.1 Aquatic Macrophytes in Lakes

The littoral zone is the interface in stream or uplands and the lake ecosystem. In many lakes, littoral regions and their associated flora are major contributors to overall productivity and biogeochemical cycles (Wetzel 1982). Submersed and emergent macrophytes grow between shore and pelagic zones and can attenuate the flow of materials from land to open waters (Carpenter and Lodge 1986).

In temperate lakes, macrophytes undergo an annual cycle of production and decomposition (Carpenter 1980; Wetzel 1982). Macrophyte productivity is typically determined by measuring fluctuations in biomass (Westlake 1964). In annual species, initial biomass of seeds is negligible. As the growing season continues, biomass increases in sigmoid fashion, reaching a plateau (usually around the time of flowering) until productivity declines as plant tissue ages, or in temperate zones, is killed by frost. Biomass decreases when respiration exceeds net production in mature plant tissues (Carpenter 1980; Wetzel 1982).

2.1.1 Aquatic Plant Establishment

Aquatic macrophytes can be classified as emergent, floating-leaved, or submersed (Wetzel 1982). Emergent macrophytes establish on hydric soils that are either saturated or submersed. Floating-leaved macrophytes have submersed roots and establish in the middle littoral zone to depths not exceeding three meters. Submersed macrophytes can occur at all depths, limited by the extent of the photic zone.
Vascular angiosperms can occur to about 10 m (2 atm hydrostatic pressure). Below these depths gas exchange becomes an insurmountable challenge. However, non-vascular macrophytes, such as bryophytes and charophytes are not constrained by water pressure, and may be present in deeper waters as long as some photosynthetically active radiation (PAR) is available (Wetzel 1982). Submersed angiosperms possess abundant intracellular lacunae for rapid diffusion of gasses (Grace and Wetzel 1978).

While emergent macrophytes are similar physiologically to the terrestrial plants they evolved from, submersed angiosperms have many adaptations to unique and challenging conditions within aquatic environments. Leaves are only a few cells thick with photosynthetic pigments concentrated in epidermal and contain almost no lignin, sclerenchyma or collenchyma (Sculthorpe 1967; Wetzel 1982). Leaves are finely divided to increase surface area, thus allowing plants to maximize exposure to light and nutrients (Carpenter and Lodge 1986; Wetzel 1982). Some submerged plants also exhibit heterophylly (vegetative polymorphism). A morphological shift exists where smaller, finely divided, submersed leaves give way to larger, entire leaf shapes that are surface floating or aerial (Judd et al 1999; Wetzel 1982). Temperature, carbon dioxide, light, and ethylene concentration mediate heterophylly (Judd et al 1999; Wetzel 1982).

Submersed plants also face problems with CO₂ assimilation. Thin cell walls, reduced cuticle and concentrated chloroplasts aid in exchange of gasses and nutrients (Wetzel 1982). Certain species of submersed macrophytes assimilate carbon through
the decarboxylation of bicarbonate ions (Carpenter and Lodge 1986; Wetzel 1982). This represents a significant adaptation in the face of reduced available CO₂.

Most non-rooted plants sequester nutrients from the water; few root aquatic plants utilize water as a source of nitrogen and phosphorus, because the relative availability is much greater in interstitial water and the soil. Interstitial waters of sediments have higher nutrient concentrations than ambient waters and are major sources of phosphorus (Bristow and Whitcombe 1971; Carpenter and Lodge 1986; Welsh and Denny 1979; Wetzel 1982). Macrophytes effect the substrate by enhancing sediment deposition and aiding in retention of detrital material (Dawson 1980).

Whether macrophyte beds act solely as nutrient sources or sinks is debatable (Carpenter and Lodge 1986). In general, stands of aquatic macrophytes are sinks for particulate matter in sedimentation and are sources for dissolved phosphorus and organic carbon (Carpenter and Lodge 1986; Wetzel 1982). On annual scales however, macrophyte stands are sources of particulate organic matter (POM). During the growing season, aquatic plants act as phosphorus sinks (Howard-Williams 1993) while during senescence they act as a net source (Landers 1982). The composition of macrophyte communities influence dissolved phosphorus and organic carbon fluxes from littoral areas (Carpenter et al. 1983; Carpenter and Lodge 1986; Wetzel 1982). Sometimes open water trophic status is inversely proportional to littoral productivity because the wetland acts as a nutrient sink to protect the open water (Carpenter and Lodge 1986).
Light availability is a major factor in the establishment and growth of aquatic vegetation. Light travelling through water is attenuated exponentially. Competition for light is the basis of the zonation with depth in macrophyte communities (Carpenter and Lodge 1986; Grace and Wetzel 1978; Peltier and Welch 1969; Twilley and Barko 1990; Wetzel 1982). Many species of plants have undergone extensive adaptation to low light and differing temperatures. Some submersed species respond to low levels of light by altering their morphology (stem length) and chlorophyll-α concentration (Barko and Smart 1981; Titus and Adams 1979; Twilley and Barko 1990). Light availability and subsequent shading within the macrophyte canopy can influence competitive success of one species over another (Twilley and Barko 1990; Wetzel 1982).

2.1.2 Interactions of Aquatic Macrophytes with other Littoral Biota

The role of aquatic plants in biological interactions depends upon plant distribution, productivity, and biomass (Carpenter and Lodge 1986). These factors are highly variable between oligotrophic and eutrophic systems. Macrophytes are colonized by many species of epiphytic fauna including microbes, algae and other consumers. Dissolved nutrient exchange of epiphytic consumers within the water is higher than that of their macrophyte hosts (Carignan and Kalff 1982; Carpenter and Lodge 1986; Howard-Williams 1981). Nutrient cycling and exchange is unique among macrophyte-epiphyte interactions. For example, organic carbon released by macrophytes is vital to epiphytic bacteria (Allen 1971). Phosphorus is not emitted by living aquatic plants (Barko and Smart 1980a; 1980b; 1981; Smith 1978; Wetzel
1982). Therefore epiphytic bacteria cannot obtain phosphorus from their host. However, as macrophytes die, phosphorus is assimilated with ease by epiphytic algae, thus providing a food source for grazers (Carpenter and Lodge 1986; Kistritz 1978).

Epiphyte food on macrophyte surfaces may be responsible for high invertebrate densities within macrophyte stands (Carpenter and Lodge 1986; Dvorak and Best 1982; Lodge 1985). Invertebrates may also use macrophytes themselves as a food source and for protection from predators (Carpenter and Lodge 1986). Most invertebrates associated with macrophytes utilize epiphytes as a food source as opposed to plants themselves (Carpenter and Lodge 1986; Cattaneo 1983; Reavell 1980; Rosine 1955). Greater surface area of aquatic macrophytes is associated with more varied and extensive invertebrate fauna (Rasmussen 1988; 1993; Rosine 1955). As macrophyte biomass increases, the associated community becomes dominated by smaller invertebrates (Mittelbach 1981). The association of dense macrophyte stands (especially *M. spicatum*) with small invertebrate species seems to be a function of size-selective predation by fishes rather than concealment within the canopy (Mittelbach 1981). For example, large invertebrates in fishless ponds (odonates, trichopterans, gastropods, leeches, and amphipods) reside primarily on macrophytes (Crowder and Cooper 1982; Morin 1984). These are the first to be eaten at high fish densities (Crowder and Cooper 1982; Mittelbach 1988; Northcote 1988; Rasmussen 1993).

Complex interactions between macrophytes, epiphytes, and associated grazers suggest symbiosis: macrophytes provide nutrients to epiphytes, which in turn protect
them from grazers by acting as a preferred food source (Hutchinson 1975). Macrophytes that release nutrients to support epiphytes are selectively favored (Wetzel 1982) and are colonized more (Carpenter and Lodge 1986). Many grazers lack sufficient feeding mechanisms to puncture plant tissue. Macrophytes may passively benefit grazers by providing substrate for epiphyte food and refuge while grazers eliminate excessive epiphytic growth that would hinder efficient photosynthesis and nutrient exchange (Carpenter and Lodge 1986).

Although food web studies of freshwater littoral zones are rare (Carpenter and Lodge 1986), much evidence suggests that herbivory of aquatic macrophytes is common. Muskrats and waterfowl for example have been shown to greatly reduce cattail (Typha latifolia) biomass (Pelikan et al. 1971; Smith and Kadlec 1985). Macrophytes are also an important food source for many species of temperate fishes. Feeding and spawning habits of grass carp (Ctenopharyngodon idella) have a devastating effect on macrophyte stands (Mitzner 1978). Tilapia sp. also feed voraciously on aquatic plants (Bowen 1982). Consumption by invertebrates, mammals, waterfowl, and fishes may be important to production, diversity and contribution of macrophytes to nutrient turnover in freshwater littoral zones (Carpenter and Lodge 1986; Wetzel 1982).
2.1.3 Introducing Littoral Flora to an Ecosystem

In reservoirs and other simple aquatic systems, wildlife managers and engineers may choose to introduce or construct littoral vegetation to improve productivity and biological diversity. Designs that use natural processes to achieve objectives are desired, although they may not develop predictably (Mitch and Gosselink 1993). Several factors require consideration if littoral vegetation construction is to be successful. These include lake level fluctuation and depth, seasonal and year-to-year pulses, inflow/outflow, basin morphology, substrate, and vegetation type (Mitch and Gosselink 1993).

System hydrology conducive to macrophyte establishment must be accompanied by sufficient nutrients in the substrate. Although the nutrient conditions necessary for optimum growth are not well defined, low levels associated with clay, sandy, or organic soils may pose problems for vegetative establishment (Allen et al. 1989). Fertilization may be necessary to promote growth in some situations but must be used cautiously as to avoid future eutrophication and macrourient sink formation (Mitch and Gosselink 1993).

Specific type and species of vegetation to be constructed depends on hydrologic condition, climate and management objective of the lake. Although some plant species are exotic or undesirable in terms of ecosystem value or aesthetics, it is important to take into account system characteristics and potential ability to support desired species. Various planting techniques include transplantation of roots, rhizomes, tubers, or mature plants. Seeding is also a viable option and is best
employed when the lake is at minimum depth (Mitch and Gosselink 1993). Vegetative mats can also be introduced to promote establishment by fragmentation (Keast 1984).

2.1.4 Undesirable Aquatic Plants

Exotic aquatic plants ("weeds") are viewed by most wildlife managers as undesirable or nuisance species. Population explosions of invasive species of aquatic plants can have undesirable consequences. Floating exotic species that clog waterways include water hyacinth (*Eichhornia crassipes* (Mart.)), water fern (*Salvinia auriculata* (Aublet)), and water lettuce (*Pistia stratiotes*) (Holm et al. 1969). Submersed exotic species include members of the genera *Potamogeton*, *Elodea*, *Ceratophyllum*, *Najas*, *Myriophyllum*, *Ranunculus*, *Hydrilla*, and *Utricularia* (Forsyth et al. 1997; Holm et al. 1969; Netherland et al. 1993; Warrington 1990). Emergent species that are problematic include members of the genera *Scirpus* (bulrushes), *Typha* (cattails), *Nymphaea* (water lilies), *Nuphar* (spatterdock), *Juncus* (rushes), *Sagittaria* (arrowhead), and *Alternanthera* (alligatorweed) (Holm et al. 1969; Warrington 1990). They can destroy fisheries, interfere with hydroelectric power generation, create irrigation and drainage problems, impede surface water use, smother shellfish beds, provide mosquito breeding sites, clog intakes, pollute municipal water supplies, and contribute to a myriad of other problems (Aiken et al. 1979; Creed 1998; Carpenter and Adams 1977; Grace and Wetzel 1978; Holm et al. 1969; Newroth 1993). Invasive species may outcompete native species for space and nutrients due to high growth rates, fecundity, and a lack of native predators.
There are currently many methods for controlling nuisance vegetation in North America. Preventative measures include boat checks, quarantines, fragment barriers, and education (Newroth 1993). The earliest efforts to control aquatic weeds were performed manually through use of hand tools to cut, dig out, or harvest the problem plants (Holm et al. 1969). Since the early twentieth century, mechanical harvesters and herbicides have been used to remove aquatic vegetation. Mechanical harvesting offers several ecological and practical advantages. It is species specific, provides immediate removal of target vegetation, removes nutrients from target systems, and avoids side effects of chemical herbicides (Brooker and Edwards 1975; Carpenter and Adams 1977; Newroth 1993).

Bottom barrier application, and suction harvesting are two more recent developments to control vegetative growth. Barrier materials (polyester geotextiles) are applied by SCUBA divers and kill vegetation through light deprivation. It has been used successfully since about 1981 (Newroth 1990; 1993). Suction harvesting is useful in areas where cutting and herbicides are unacceptable, i.e. public water supplies. This method requires removal of plant material by a diver using a suction tube. Spoil is collected in a wet well where water and sediment are allowed to escape while plant material is retained (Eichler et al. 1993).

Habitat manipulation is a viable management solution in reservoirs in particular. Methods include scheduled water level changes, sediment modification / amendment in the form of lime application or alum flocculation, and chemical shading (water column dyes) (Newroth 1993). Experimental methods include the use
of lasers and ultrasound to destroy plant material or limit a plant’s metabolic capability (Newroth 1993).

Many selective herbicides have been developed to control and eradicate aquatic weeds. Some are quite specific while others are broad-spectrum defoliators (Holm et al. 1969). In the past, large quantities of toxic chemicals were employed indiscriminately with drastic consequences to surrounding flora and fauna. Herbicides are now tested extensively for efficacy, potency, toxicity, and biological magnification implications (Farone and McNabb 1993). Herbicides formerly used include 2,4-D (2,4-dichlorophenoxyacetic acid), Diquat (6,7-dihydridipyrido [1,2-α: 2′, 1′-c] pyrazidiinium dibromide), amitrole-t (3-amino-1, 2,4-triazole + ammonium thiocyanate) and Acrolein (acrylaldehyde) (Holm et al. 1969). Today, although the former agents are still widely used, new agents such as linuron [8-(3,4-dichlorophenyl)-1-methoxy-1-methyl urea], fluridone (1-methyl-3-phenyl-5-[(3(trifluromethyl) phenyl)-4 (IH)-pyridinone), glyphosphate, thidiazuron, and atrazine are used extensively (Christopher and Bird 1992; Forsyth et al. 1997). Atrazine is the most widely used herbicide in the United States (Christopher and Bird 1992). These agents affect vegetation in a variety of ways including inhibition of respiration, photosynthesis, and amino acid biosynthesis as well as cytokinin interference (Christopher and Bird 1992).

Biocontrol strategies are centered on the search for biological limiters of invasive species in their native habitats. In their native habitat, most species do not
present infestation problems. This is due to presence of bacteria, fungi, disease, predation, or other selective pressures within native habitats (Holm et al. 1969; Creed 1998). Biocontrols utilized to combat aquatic plants include introduction of the herbivorous snails *Marisa cornuarietis* and *Pomacea australis*. Herbivorous fishes used to combat vegetation include the common carp *Cyprinus carpio*, grass carp *C. idella*, and *Tilapia sp.* (Holm et al. 1969). The herbivorous weevil *Euhrychiopsis lecontrei* has been shown to be particularly effective in reducing standing crops of milfoil (Creed 1998; Creed and Sheldon 1993). Butterfly larvae, especially *Acentria ephemerothella* have also been associated with invasive vegetation declines (Johnson et al. 1997).

Nuisance vegetation can also be controlled by competition from the introduction of more desirable species (Newroth 1993). Biological controls, although sound in premise, are potentially dangerous methods of nuisance vegetation eradication. Upon introduction, control species are no longer subject to the same stresses that held them in check in their native environments. An example of this is the Chinese grass carp (*C. idella*), a species that can totally eliminate all aquatic and some riparian vegetation.

### 2.1.5 Eurasian Watermilfoil

A major contributor to nuisance vegetation in the United States is milfoil (Figure 1.). Eurasian watermilfoil (milfoil) is an extraordinarily successful submerged aquatic macrophyte in terms of distribution, fecundity and competitive ability
milafoil was first described from Europe by Linnaeus in 1753 and was present in North America in the Chesapeake Bay area by the early 19th century (Grace and Wetzel 1978). milfoil is a rooted perennial angiosperm with long resilient stems and finely divided leaves. The plant may grow to lengths in excess of 4 m. Flower parts occur as a spike and must emerge prior to fruit formation and maturity (Patten 1954). Leaves occur in whorls of four with 10 to 26 paired leaf divisions (Patten 1954). As with most submerged macrophytes, leaves of milfoil are covered with a very thin cuticle (Judd et al. 1999; Sculthorpe 1967). Essentially vestigial stomata occur on leaf surfaces as well as hydropoten for ion absorption (Grace and Wetzel 1978). Chloroplasts are concentrated in epidermal tissue as an adaptation to aquatic life (Grace and Wetzel 1978; Wetzel 1982). The root system of milfoil is adventitious in nature, forming abundant root hairs. Like most submerged macrophytes, the vascular system of milfoil is simplified and there is little xylem tissue (Sculthorpe 1967). Phloem tissue is analogous to that of most land plants (Grace and Wetzel 1978). milfoil possesses advanced lacunae and abundant aerenchymous tissue to serve as gas reservoirs to facilitate diffusion between roots and shoots as well as to provide buoyancy (Grace and Wetzel 1978). These tissues can account for up to 43% of total volume in mature plants (Hartman and Brown 1967).

milfoil is a highly fecund species, and is capable of reproduction through several strategies. milfoil propagates sexually by seed, asexually by fragmentation, abscission, or stolon formation, and can overwinter as an evergreen or in the form of
a dormant apex attached to a stoloniferous rhizome or rootstock (Grace and Wetzel 1978). Viable propagule formation requires emergence of a flowering spike in this species (Patten 1954). Seed production is important in both dispersal and survival of periodic drying. Pollen is typically transferred by wind (Grace and Wetzel 1978). Stigmas open before stamens in order to promote outcrossing (Aiken et al. 1979). Establishment of seedlings is the most fragile stage of the milfoil life cycle (Grace and Wetzel 1978). Fragmentation probably occurs accidentally resulting from wind or wave action, surface water use or mechanical harvesting. Fragments are produced when a parent plant releases a portion of itself. Abscissions typically develop roots prior to release to improve establishment success (Grace and Wetzel 1978). Fragmentation and abscission seem to be the most efficient and likely means of dispersion within a body of water (Grace and Wetzel 1978).

milfoil typically concentrates most of its biomass as a dense canopy or mat near the water surface thus giving the illusion that it is more productive than it actually is (Grace and Wetzel 1978). This growth form allows it to outcompete smaller species for available light (Aiken et al. 1979; Grace and Wetzel 1978; Nichols and Rogers 1997; Wetzel 1982). Furthermore, milfoil appears to be quite shade tolerant and possesses a low CO₂ compensation point (Grace and Wetzel 1978). Free CO₂ appears to be the preferred form of dissolved inorganic carbon for this species (Grace and Wetzel 1978). Endodermal root tissues of milfoil possess xylem in the form of a casparian strip that aids in mineral absorption (Sculthorpe 1967). This adaptation combined with abundant root hairs for added surface area, enables this
species to be quite metabolically efficient (Grace and Wetzel 1978; Shannon 1953; Westlake 1975).

Milfoil is highly successful in colonizing new habitats. It can outcompete other submersed macrophytes as it is shade tolerant and highly efficient at inorganic carbon fixation (Sculthorpe 1967; Grace and Wetzel 1978). Milfoil is well adapted for nutrient uptake from sediments and has low nitrogen and phosphorus requirements (Barrett et al. 1993; Gerloff and Krombholz 1966; Grace and Wetzel 1978; Smith 1978). Anatomy, growth form, metabolism, high fecundity, and resistance to low nutrient concentrations are major factors allowing milfoil to successfully compete for and establish within new habitats (Nichols and Rogers 1997; Titus and Hoover 1991).

Milfoil has become a considerable nuisance in rivers, ponds, lakes, and wetland areas throughout North America (Aiken et al. 1979; Carpenter and Adams 1977; Carter and Rybicki 1994; Creed 1998; Eichler et al. 1993; Grace and Wetzel 1978; Newroth 1993). Although it was first reported in North America in the early 19th century it became widely distributed in the United States by the 1960's (Couch and Nelson 1986). Milfoil is currently present in at least 41 states and 3 Canadian provinces in North America (Creed 1998; Nonindigenous Aquatic Species Database 1997). Milfoil behaves similarly to other nuisance species, presenting significant challenges to maintenance of water quality and quantity within reservoirs. Problems include direct blockage of flow and flooding, loss of reservoir storage capacity, water loss due to evapotranspiration, pump interference caused by fragmentation, flow measurement interference, sedimentation increase, bank erosion,
and canal distortion (Aiken et al. 1979; Carpenter and Adams 1977; Grace and Wetzel 1978; Newroth 1993; Smith and Barko 1990). Dense mats impede surface water use for fishing and recreation.

Reduced water quality can be associated with various human health impacts. milfoil infestation can have adverse effects on aquatic habitat components such as oxygen, pH and carbon dioxide fluctuation (Grace and Wetzel 1978), sub-minimum dissolved oxygen content (below 4 mg/l; Wetzel 1982), and high surface pH (>9.5) for fish (Newroth 1993; Pauley and Thomas 1988). Dense plant canopies can cause anaerobic sediment phosphorus release from sediments without an oxidized microlayer (Newroth 1993; Smith 1978; Wetzel 1982). milfoil can have adverse effects on aquatic life as well. milfoil may grow on benthic spawning areas, thus impeding reproductive success of some fishes (Newroth 1990). Dense populations can outcompete and displace some native plant species that may be preferred habitat for fish and waterfowl (Grace and Wetzel 1978; Newroth 1990; 1993; Nichols 1991). Organic compounds (phenols) released from milfoil may be allelopathic (Newroth 1993). Dense submerged vegetation provides habitat for vectors of human diseases including mosquitoes, blackflies, and aquatic snails (Newroth 1993).

milfoil is managed utilizing several strategies. Total elimination of an established population without devastating ecological consequences to the management site is neither possible nor practical. Primary management is prevention of spread. If discovered in early life cycle stages (prior to rootstock formation and fragmentation) careful hand removal of all plant material and roots is effective.
Application of sediment barriers to affected areas will prevent regrowth of overlooked fragments (Newroth 1990; 1993). This strategy requires extensive and regular monitoring so as to detect milfoil introduction. This strategy is effective in lakes with high recreational value and justifies survey costs (Warrington 1990).

When milfoil has become established however, different strategies must be employed. Various sediment-tilling operations in the interest of root removal are aesthetic maintenance programs (Carpenter and Adams 1977; Holm et al. 1969; Newroth 1993). These methods are slow, expensive, cause fragmentation and spread, and should be considered only when established populations are already present throughout the lake (Warrington 1990). Herbicides such as fluridone and 2,4-D have been shown to be useful in selective elimination of root crowns of milfoil (Christopher and Bird 1992; Farone and McNabb 1993; Holm et al. 1969; Netherland et al. 1993; Newroth 1993). Although the EPA has approved both chemical herbicides for aquatic use, fluridone is preferred over 2,4-D as it is not carcinogenic (Hamelink et al. 1986), demonstrates low toxicity to vertebrates and aquatic invertebrates, and is effective at lower concentrations (Netherland 1993). Herbicides must be employed cautiously. Nonspecific macrophyte removal is detrimental to aquatic environments, and selection of a proper management and control strategy depends upon lake size and intended use.

Although prevention of spread is the most cost-effective control option for this species, it seems impractical to inspect all boats and trailers at all launches on large
lakes and reservoirs (Warrington 1990). Boat cleaning, public awareness and education are required to prevent infestation (Warrington 1990).

2.1.6 Role of Aquatic Vegetation in Lake Ecosystems

It is widely understood that aquatic vegetation can present challenges if certain species are allowed to establish and proliferate without control. It must not be overlooked however that aquatic macrophytes possess many beneficial qualities that make them assets to ecosystems. Moderate density and patchy distributions of aquatic vegetation are essential to ecosystem productivity and biological diversity (Adams and McCracken 1974; Anderson 1984; Carpenter and Lodge 1986; Cyr and Peters 1996; Downing and Plante 1992; Hinch and Collins 1992; Keast 1984; Newroth 1993; Randall et al. 1996; Rosine 1955; Wilcox and Meeker 1992).


Aquatic macrophyte beds in lake littoral zones benefit aquatic fauna. Submersed plants provide shelter, nesting sites, hunting grounds, and forage to a

In light of the biological value of moderate aquatic macrophyte densities, it stands to reason that oligotrophic, man-made systems such as reservoirs and other multipurpose impoundments would benefit greatly from their presence. Addition of any aquatic plants in waters devoid of such would be a vast improvement from an ecological standpoint. Likewise, macrophyte introduction within these systems, particularly milfoil, has been shown to improve habitat for fishes and their invertebrate prey (Adams and McCracken 1974; Jude and Pappas 1992; Keast 1984).

2.2 Importance of Littoral Zones to Fish Communities

Aquatic macrophytes are of particular importance to fishes within the littoral zone of freshwater lakes. Aside from providing suitable reproductive habitat for phytophilic fish species, they provide cover and protection from predation, and a highly productive feeding environment by supporting rich invertebrate communities (Cyr and Downing 1988a,b; Downing and Plante 1993; Johnson et al. 1997; Keast 1984; Lodge 1985, 1991; Lodge and Lorman 1987; Mills et al. 1981; Randall et al. 1996). It has been shown that vegetated sites have higher associated fish densities,
biomass and species richness than do nonvegetated sites (Keast et al. 1978; Randall et al. 1996). Relationships between macrophytes and fishes within lentic systems are unique depending on geographic location, and plant and fish species composition (Killgore et al. 1993; Randall et al. 1996). Macrophytes increase structural complexity within littoral zones, increasing fish species richness (Crowder and Cooper 1982; Eadie and Keast 1984; Randall et al. 1996).

In addition to vegetation, fish production may be related to other factors. Randall et al. (1996) demonstrate total phosphorus, littoral basin morphology, effective lake fetch and wind exposure to correlate positively with fish production. Total phosphorus correlation is consistent with findings by Downing et al. (1990), Downing and Plante (1993), Hanson and Leggett (1982) and Randall et al. (1993). Findings are consistent at population and community levels. Downing and Plante (1993) correlate fish production with chlorophyll-α concentration, temperature and pH as well. This suggests a ‘bottom-up’ trophic cascade within freshwater lakes (Downing and Plante 1993; McQueen et al. 1989).

Littoral basin morphology is important because the slope of the littoral zone will determine extent and dynamic nature of the habitat (Randall et al. 1996). Slope may influence convective currents and likewise have an effect on renewal rates of littoral waters (Horsch and Stefan 1988). Fetch and wind exposure are important considerations as they influence wave energy, wind currents and convective currents that impact littoral zones (Randall et al. 1996; Wetzel 1982). Exposed shorelines and points may be less than suitable sites for macrophyte establishment due to increased
possibility of physical damage from wave action and increased turbidity (Randall et al. 1996; Wilson and Keddy 1986). Wave action can positively impact macrophytes at intermediate levels of exposure by decreasing suspended solid deposition on plant surfaces and aiding photosynthesis (Keddy 1983; Scheffer et al. 1992; Wetzel 1992).

Changes in littoral productivity ultimately influence fish distribution (Randall et al. 1996). High densities and species richness occur at intermediate rates of disturbance. Therefore fish productivity is equally subject to mechanical forces at work on macrophyte communities (Nixon 1988; Randall et al. 1996). Aquatic macrophytes and detritus comprise the organic substrate foundation vital to higher trophic levels.

2.3 Muskellunge in Reservoirs

In lakes valued for recreation, fisheries management is a high priority from both an ecological and economic standpoint. Sport fishermen prize muskellunge (Esox masquinongy) because of both its disposition and niche within lake ecosystems (Figure 2). Tremendous effort is usually expended for angling success. Authenticated records indicate a maximum length exceeding 1.75 m and weight approaching 30 kg, although these values are hardly common (Etnier and Starnes 1993). Sportfishing popularity has led to extensive stocking programs in many states (Axon 1981; Dombeck et al. 1984; Johnson and Margenau 1993).

Muskellunge are native to the Ohio River basin (including Cumberland and Tennessee Rivers), the upper Mississippi River basin, the Laurentian Great Lakes,
Figure 2. *Esox masquinongy* Mitchill. (Etnier and Starnes 1993).
southern Hudson Bay tributaries, and few North Atlantic coastal drainages (Etnier and Starnes 1993).

Muskellunge biology is similar to that of other esocids such as pikes and pickerels. Spawning occurs later for muskellunge than other species because the preferred water temperatures are between 9.4 and 15° C (49-59° F) in April or May (Dombeck et al. 1984; Etnier and Starnes 1993). Optimum spawning habitat appears to be 1 m deep pools in streams with flow rates between .6 and 1.2 m/km with dense cover (Etnier and Starnes 1993). Spawning behavior is associated with upstream migration into upper reaches, typically upper 12-17% of inhabited stream area (Etnier and Starnes 1993). Post spawning migration is common, in some cases to distances up to 20 km (Etnier and Starnes 1993). A single female may cast over 180,000 eggs on the substrate (Etnier and Starnes 1993). Muskellunge exhibit no parental care, thus eggs are vulnerable to siltation in turbid waters (Dombeck et al. 1984). Eggs of *E. masquinongy* are non-adhesive and maintain contact with material on stream substrates (Hess and Heartwell 1978).

Muskellunge are top predators in their environment’s food web. Recently hatched muskie feed on zooplankton; however, they become picivorous after only four days, utilizing fish as a primary food source for the remainder of their life. Adult muskellunge are large, aggressive ambush predators that feed on a variety of fishes
including herring, suckers, shad, perch and even other muskellunge (Etnier and Starnes 1993; Johnson and Margenau 1993).

In native habitats, hybridization occurs between muskellunge and northern pike (Esox lucius) and these individuals are commonly referred to as tiger muskie (E. masquinongy x E. lucius) (Etnier and Starnes 1993). Muskellunge average growth is approximately 15 cm per year. Individuals are sexually mature between the third and fourth years. Life span of this species is highly variable. Few members of transplanted or stocked populations were thought to live beyond 6 years (Etnier and Starnes 1993). In northern populations however, life span estimates exceed 25 years (Etnier and Starnes 1993).

Muskellunge are an important and popular gamefish in North American waters. Many lakes and reservoirs are managed to promote this species and its prey in the interest of angling. As many of these waters are not native habitats, populations are sustained through artificial propagation and stocking programs (Axon 1981; Bimber and Nicholson 1981; Dombeck et al. 1984; Johnson and Margenau 1993). Natural spawning and subsequent reproductive success are rare in many reservoirs and impoundments (Bimber and Nicholson 1981; Hess and Heartwell 1978). Human activities including damming streams, draining wetlands and managing water levels with large fluctuation have contributed to significant declines of muskellunge spawning and nursery habitat (Dombeck et al. 1984). Subtle environmental changes can be highly detrimental to reproductive success (Bimber and Nicholson 1981). Drastic disturbances associated with water level fluctuations in reservoirs
significantly impact egg and fingerling mortality, and all but eliminate the possibility for natural spawning success within these waters (Dombeck et al. 1984; Hess and Hartwell 1978). Silt and detritus often cover non-adhesive eggs broadcast by spawning females within reservoirs, reducing dissolved oxygen content where substrate and water meet (Dombeck 1979).

Establishment of stable muskellunge sport fisheries has become the responsibility of hatcheries. Management strategies for muskellunge focus on annual harvest by anglers, harvest regulations, and stocking programs. As muskellunge are typically stocked at a rate of one fingerling per acre annually, each year class represents a significant investment (Dombeck et al. 1984). Hatchery production of this species may become a less efficient management option due to difficulties with raising this species and high cost. Further, survival of hatchery stock to legal size (36-42 in. dependent upon individual state regulation) is usually 10% (Johnson 1978). A supplement to fingerling stocking programs could be introduction of artificially fertilized eggs to selected substrates (Dombeck et al. 1984). This technique is promising, particularly in areas where ideal spawning substrates are not available. Identification and conservation of natural muskellunge spawning habitat will be critical to maintaining supplies for hatcheries (Dombeck et al. 1984, Johnson 1978, Hess and Heartwell 1978).
3. Materials and Methods

3.1 Study Site

Cave Run Lake is an 8,270-acre (3,319 ha) impoundment of the Licking River located in Menifee, Morgan, Bath, and Rowan Counties in Eastern Kentucky, extending 38 miles above a dam (Figure 3). It is situated at the edge of the Cumberland Plateau in the Eastern Coal Fields physiographic region. Most of the reservoir and its watershed (73%) are within the Daniel Boone National Forest (Buynak et al. 1986; Luken and Bezold 2000). The regional climate is temperate with an average daily maximum temperature of 20.6°C; average annual total precipitation of 116.6 cm. Pennsylvanian and Mississippian age materials are exposed around steep slopes. Sandstone, siltstone and shale layers are principal components of the shoreline matrix (Luken and Bezold 2000). Principal soiltype along 267 km of shoreline is defined as Cranston Gravelly Silt Loam (Avers et al. 1974; Luken and Bezold 2000).

Designed as a flood control reservoir and constructed in 1974, Cave Run Lake encompassed 46.38 miles of Licking River and surrounding drainage, 10 miles of North Fork, and 6.35 miles of Beaver Creek (Axon 1981). Cave Run Lake possesses an upper riverine zone that is narrow at its origin and grades into a broad transitional zone. This transitional zone eventually opens into a large basin (Luken and Bezold 2000). Water exchange rates for Cave Run Lake rarely exceed 105 days (Axon 1981). The reservoir is drawn down in late autumn and early winter to act as a buffer for rainfall during late winter and early spring. Water level regulation on this reservoir is
Figure 3. Large Map: Cave Run Lake, Kentucky.
frequently imprecise, resulting in lake levels exceeding ideal summer pool (223 m mean sea level). Cave Run has a maximum depth of 29.5 m and an average depth slightly below 9 m. Buynak et al. (1989) used Carlson’s trophic state index (TSI) to classify the lake basin’s trophic state. Chlorophyll-α concentrations collected by the U.S. Army Corps of Engineers indicate that upper regions of the lake are eutrophic (TSI, 56). Middle and lower lake regions are mesotrophic and oligotrophic respectively; (TSI, 41) and (TSI, 38) (Buynak et al. 1989). However, in 1994, Davis and Reeder (2001) found the chlorophyll-α TSI suggested the lake was oligotrophic along its entire length.

3.2 Aerial Photography

Infrared aerial photographs of the littoral zone of Cave Run Lake were taken in September of 1999 and again in September of 2000. Vegetation was mapped by sight during the 2000 flyover. As the aircraft followed its planned flight path, a scale map of Cave Run Lake proper was oriented coincidentally with heading azimuth using a Brunton Eclipse® model 90 compass. Macrophyte location and density were physically drawn on the map as photographs were taken simultaneously. Thirty-five photographs were taken during the flyover in 1999, and eighteen photographs were taken in 2000 from an altitude of 420 m at an angle closely approximating 90°. Tim Holbrook using a Nikon 35mm camera equipped with a yellow filter and Kodak EIR 135 Infrared film performed all photography. Shudder speed aperture was set at 250
Figure 4. Map of Cave Run Lake with icons demonstrating sampling sites for the present survey. Triangles = soil sample sites; squares = water sample sites; circles = insect sample sites; pentagrams = plant sample sites. Multiple horizontal icons indicate multiple sample types at one site.
with subfocus set at infinity. All film underwent AR-5 processing at Rocky Mountain Film Laboratories in Aurora, Colorado.

3.3 Water Quality Measurements

Duplicate 500 ml samples were taken from the approximate center of the water column using a Van Dorn sampler at 29 sites (Figure 4). Water samples were examined for total nitrogen (NO₃, NH₄), total phosphorus, chlorophyll-α, sulfur (SO₄), and iron (Fe³⁺) in the laboratory. At the sample site, samples were filtered through a precombusted 0.45 µm-glass fiber filter (Whatman GF/A) directly after sampling. The filter was placed in a centrifuge tube, wrapped in aluminum foil, and placed on ice. In the lab, 10 ml of 90% alkalized acetone was added to the filter, the filter was homogenized, then left to steep at < 4°C for at least 24 hours. The tube was centrifuged, and the extract examined for chlorophyll-α using a Turner Model 10AU Fluorometer (Turner Designs, 845 W. Maude Ave., Sunnyvale, CA 94086) (EPA Method 445.0) (Kopp and McKee 1979). The fluorometer was standardized using NIST standards purchased from Turner Designs, and checked against standards made using chlorophyll-α from *Anacystis nidulans* (Sigma Chemical Co., P.O. Box 14508, St. Louis, MO 63178). These were also checked against Turner gel standards.

Concentrations of soluble reactive phosphorus in filtrant were analyzed using the ascorbic acid method (EPA Method 365.3) (Kopp and McKee 1979). Ammonium was determined via Nesslerization (EPA Method 354.1) (Kopp and McKee 1979). Nitrate was determined by using the sulfanilamide method after copperized cadmium reduction (EPA Method 353.3) (Kopp and McKee 1979). Iron was determined using
the phenanthroline method (APHA 1985). Sulfate concentration was determined using the turbidimetric method (EPA Method 375.4) (Kopp and McKee 1979).

All spectrophotometric analyses were performed using a Hach DR 2010 or 2000 Spectrophotometer (Hach Chemical Company, P.O. Box 389, Loveland, CO) fitted with a flow-through cell to eliminate problems with matched glassware. Two sets of standards were analyzed concurrently with the samples to construct a standard curve. For each chemical analyzed, a set of samples was analyzed. Standard curves always included at least three samples inside the range of concentrations measured, and two samples at the extremes of the concentrations measured. The two sets of standards were made by diluting standards purchased from two separate suppliers (Hach Chemical Co. and Fisher Scientific).

A depth profile was obtained (readings taken at one meter intervals) using Hydrolab® monitoring instruments (Datasonde® 3, Datasonde® 4, and Scout® 2). All instruments were calibrated prior to field sampling. Profile data include temperature (°C), conductivity (microsiemens), salinity (mg/L), dissolved oxygen (mg/L), PAR, and pH for each sample site. The probe was calibrated within 24 h of sampling. Dissolved oxygen was air calibrated based upon barometric pressure, pH was calibrated using 4.00, 7.00, and 10.00 standards, temperature using an NIST thermometer, and conductivity using NIST standards (all standards from Fisher Scientific, Pittsburgh, PA).
3.4 Substrate Analysis

3.4.1 Sample Collection

A total of sixteen soil samples were taken from Cave Run Lake in both weeded and non-weeded zones. Nine were taken during November of 1999 and seven were taken during the summer of 2000. November samples were obtained using an Echmann Dredge. Within weeded and non-weeded zones, the dredge was dropped, closed and retrieved. Dredge spoil (sediment) for each site was placed in a plastic bag, marked and returned to the laboratory.

Summer 2000 samples were obtained using a PVC core sampler. In weeded and non-weeded zones, a ten foot section of standard PVC plumbing pipe of 3-inch uniform diameter was driven into the sediment with a twelve pound sledge hammer. The free end of the pipe was fitted with a teflon tape sealed removable cap. After the pipe was driven to a desirable depth (30 cm) the cap was screwed on to maintain a vacuum. The pipe was then retrieved by hand and contained both sediment and the water column above it. A pipe wrench was used to loosen the cap and the contents of the sampler were allowed to filter through a five-gallon bucket with 3mm mesh in place of a solid bottom. This allowed for both retention of sediment and water escape. All soil samples were allowed to air dry and analyzed in the laboratory for total nitrogen, total phosphorus and total organic matter.
3.4.2 Nutrient Analysis

Nitrogen and phosphorus concentrations within sediment samples were determined by sulfuric acid-hydrogen peroxide digestion (HACH® Digesdahl® Digestion Apparatus Manual 1999). Post-digestion analysis for total phosphorus was performed using the ascorbic acid method (Murphy and Reily 1962). Total Nitrogen was determined by the Nessler method for ammonia (APHA 1985). Total organic matter data (including carbonates) were determined by loss on ignition (Dean 1974). Crucibles were placed in a muffle furnace at 550°C until constant weight was obtained on an analytical balance. Three grams of sediment from each of 16 samples were placed in three crucibles (one gram per crucible) and weighed to the nearest hundredth of a gram. All samples were placed in a muffle furnace at 1200°C for twelve hours. All crucibles were weighed after cooling to determine average loss on ignition per sample. Bioavailable N, P, Ca, and K were determined using a Malich III extraction on dried soils by the University of Kentucky Extension Laboratory.

3.4.3 Benthic Insect Analysis

Aquatic insects were sampled at 10 sites (Figure 3) during February and July of 2000 using a PVC core sampler. All samples were filtered and examined under a dissecting microscope for insects.

3.5 Aquatic Plant Collection and Analysis

Aquatic plants were sampled at nine sites during July of 2000 (Figure 3). Three samples were collected at each of three increasing depths. “Shallow” depth was
approximately 1 m, “medium” depth was approximately 2.5 m and “deep” depth was approximately 3.5 m. All samples were collected by hand with the aid of SCUBA.

Plant material was collected from three random 0.0625 m$^2$ plots at each depth zones at each site for a total of 27 samples. Mason jars were desiccated in a drying oven for 12 hours and preweighed on an electronic balance to the nearest hundredth of a gram. Plant samples were loaded and placed in a drying oven at 110°C for 96 hours. Samples were removed and weighed on an electronic balance to the nearest hundredth of a gram to determine mass dry weight.

3.6 Data Analysis

Unpaired t-tests were used to compare all data grouped according to vegetative cover (weeds vs. no weeds). Regression analysis and analysis of variance were used to determine relationships among data grouped according to depth.

Statistical analyses were performed using Statview® version 4.0 for Macintosh.
4. Results

Of the eighteen infrared photographs taken during the 2000 flyover, only two were shot at the angle necessary to accurately reflect the size of the lake. The rest were too oblique. Photos taken in 1999 were at the proper angle. However, IR images from both years did not reveal submersed vegetation as well as visual observation. Subsequently, the majority of these photographs were unusable for the purpose of generating a digitized, infrared vegetation map based on aerial photographs. Spatial distribution of aquatic macrophytes was estimated during the September of 2000 aerial photography session. Vegetation was mapped by sight according to relative density from an altitude of 420 m (Figure 5). Based upon these maps, approximately 27.7% of the lake was vegetated. Cave Run Lake fishing guides commented that this is the greatest extent of vegetation observed in the reservoir.

Four species of plant were observed. Shallow (<1 m) rocky sediments were dominated by water niad (*Najas minor*). Isolated patches of pondweed (*Potamogeton*) and narrow-leaved duck potato (*Sagittaria graminea*) were also present. Plant biomass in Cave Run Lake is dominated by milfoil. Depth distribution data show that plant biomass is highest at intermediate depths. At all nine sites sampled, macrophyte dry mass (g) was highest between 1.5 and 4 meters (Figure 6a). When all 27 plant samples are plotted against depth, a similar pattern is evident (Figure 6b).
Figure 5. Vegetation density map of Cave Run Lake, Kentucky; September 2000. Black areas indicate high density; gray areas indicate medium density; light gray areas indicate light density. Lake surface approximately 27.688% vegetated.
Figure 6a. Mean plant biomass vs. depth at site (all sites denoted by icon).

Figure 6b. Mean plant biomass vs. depth (all samples).
Figure 7a. Average photosynthetically active radiation (PAR); weeded versus non-weeded zones (* indicates significance α=.05).

Figure 7b. Average water conductivity; weeded versus non-weeded zones (* indicates significance α=.05).
PAR and specific conductivity were significantly higher in non-weeded zones compared to weeded zones (Figure 7a, 7b). No significant differences existed for any other water quality parameters between zones. However, some non-significant trends did exist between weeded and non-weeded zones. Nitrate (NO$_3$) concentrations are somewhat higher across weeded and non-weeded zones while ammonium nitrogen (NH$_4$) concentrations are slightly lower across zones (Figure 8a). Soluble reactive phosphorus (SRP) is higher within weeded zones (Figure 8b), as is pH (Figure 8c). Higher SRP values within weeded zones may be due to sequestered phosphorus present in interstitial waters surrounding macrophytes. Higher pH results as plants remove CO$_2$ from surrounding waters. No significant difference in soil composition existed between weeded and non-weeded zones across all parameters. Non-significant trends in total organic matter, total soil phosphorus and total soil nitrogen were present however. Total organic matter tended to be higher within weeded zones (Figure 9a). Likewise, total soil N and total soil P were higher within weeded zones (Figure 9b). These trends may be accounted for by actual plant biomass taken up in core samples within weeded zones.

No aquatic insects were found in core samples from February or July of 2000. Extensive microscopic examination of 16 sediment samples revealed a few insect and nematode fragments in sediment samples from weed beds at Scott Creek and Clay Lick in February of 2000. Cave Run Lake therefore appears to be unsupportive of benthic insects.
Figure 8a. Average soluble nitrogen trends; weeded versus non-weeded zones.

Figure 8b. Average soluble reactive phosphorus concentration trends; weeded versus non-weeded zones.

Figure 8c. Average pH trends; weeded versus non-weeded zones.
Figure 9a. Average soil organic matter composition trends; weeded versus non-weeded zones.

Figure 9b. Average total soil nitrogen and phosphorus composition trends; weeded versus non-weeded zones.
5. Discussion

5.1 Aerial Photography

Photographs taken during September 2000 were inconclusive. Aquatic macrophyte density and location were unobtainable due to limitations of the aerial photography technique. As a result, no precise comparison can be made between September 1999 and September 2000 photographs. Using aerial imagery as the primary detection method for submerged macrophytes is difficult. Although a visual observer in an aircraft can distinguish the extent of submerged weed growth, film cannot achieve resolution. It is not uncommon for aerial photographs to "miss" submersed vegetation. Thorough surface-level observations coupled with data on the aquatic environment (i.e. turbidity and wave action) are necessary for photographic method to be effective (Ferguson et al. 1993). Correct orientation, light reflection, wave action and survey altitude contribute to errors in identifying the position and density of aquatic vegetation (Ferguson et. al 1993). Different technologies may need to be applied in the future—such as satellite infrared imagery. Visual inspection was more successful in revealing the location and density of aquatic vegetation than remote sensing technology. Hand drawn maps made from visual observations are useful in determining the approximate percentage of vegetative cover based on spatial distribution; however they have a degree of subjectivity, thus limiting some of the data analysis. We found that low altitude high-resolution color photographs may be more useful than infrared images.
5.2 Plants

Although there are probably more species, there is an overwhelming dominance by one species of vegetation in shallow waters of Cave Run Lake. In our extensive surveys, we only found four species of submersed vegetation in Cave Run Lake. The three less prevalent species of plant observed were water naiad (N. minor), present only within shallow (<1m) rocky sediments, and isolated patches (Potamogeton, S. gramineae) in 0.5-1.5 m of water. Cave Run lacks sediment organic matter and associated nutrients, and experiences drastic water level fluctuation to fulfill its flood control mission in a steep landscape. These factors prove challenging for establishment of native aquatic plant populations (Carpenter and Lodge 1986; Wetzel 1982). Previous to the milfoil invasion, Cave Run Lake never established a large population of littoral vegetation. Over the 1980s and early 1990s low densities of water smartweeds (Potamogeton sp.) and Najas appeared in the more protected shallow zones. In 1996, milfoil began to establish in larger concentrations. If milfoil had not been introduced, the lake would probably still lack sufficient littoral flora to help fish reproduction. In Cave Run Lake, milfoil appears to dominate the littoral zone, concentrating its biomass between depths of 1.5 to 4 meters. These findings are consistent with other studies of milfoil (Grace and Wetzel 1978). Najas minor is not particularly shade tolerant and is most successful in waters <1 meter deep (U.S. Department of Agriculture1971). As milfoil prefers intermediate depths (1-4 m) Najas minor has yet to be excluded from this ecosystem. Patches of Potamogeton and S. gramineae are present only in locations that milfoil has yet to exploit as these two
species prefer much the same habitat (U.S. Department of Agriculture 1971). It is plausible that milfoil reproductive capability and low nutrient tolerance will eventually prove too effective a strategy to allow for coexistence between it, *Potamogeton* and *Sagittaria gramineae* within this system.

### 5.3 Water Quality

The presence of significantly higher PAR within non-weeded zones is accounted for by light obstruction by plant material. Milfoil canopy formation attenuates available light and contributes to the plant’s ability to outcompete smaller species (Aiken et al. 1979; Wetzel 1982; Grace and Wetzel 1978). In this way, milfoil beds in Cave Run Lake behave typically in their ability to control one of the major limiting factors in aquatic systems (Wetzel 1982; Grace and Wetzel 1978). Specific conductance within low salinity (<50mg/L-1) fresh waters is subsequently low (<100 µmhos cm-1). It is unlikely that soil runoff in a mostly forested landscape would create significantly higher nutrient loads to sediments of non-weeded zones. Likewise, dense macrophyte beds prevent runoff deposition much beyond their extent (Carpenter and Lodge 1986). As non-weeded zones demonstrate low sediment organic matter and carbonate composition by loss on ignition (average <60mg OM/g sediment) it is unlikely that a significant nutrient contribution was made to the surrounding waters by sediments. If anything, the presence of abundant macrophyte biomass should justify higher specific conductance within interstitial waters rather than the converse. Overall, nitrate concentrations appear higher than ammonium nitrogen concentrations across both weeded and non-weeded zones. This may be due
to preference by milfoil for NH₄ as its primary source of bioavailable nitrogen (Nichols and Keeney 1976). Somewhat higher SRP values occur within waters around weeded zones. This may be due to dissolved phosphorus released by decaying macrophyte material from the previous year (Carpenter and Lodge 1986). It is unlikely enough phosphorus to make a significant difference between zones would be released by milfoil: less than 5% of P uptake is usually released (Carpenter and Lodge 1986). The majority of littoral zone available P is bound in milfoil biomass. Higher pH levels were observed within macrophyte beds because the extraordinary photosynthetic activity of dense milfoil stands is removing CO₂ from the surrounding waters, hence increasing pH. No other littoral zone water chemistry components demonstrated a significant difference or noteworthy trend. It would appear that subtle changes in the chemical composition of littoral zone waters have little effect on milfoil establishment. However, milfoil’s ability to control available light may have profoundly negative effects on other plant species competing for the same space.

5.4 Sediment Composition

No significant differences between weeded and non-weeded zones existed across all tested sediment parameters; therefore it is difficult to determine the role of sediments in vegetative establishment within Cave Run Lake. It could be that the plants have not been established long enough to create statistically significant differences. We found some trends between zones in total organic matter, total soil nitrogen, and total soil phosphorus. Higher total organic matter and nutrients within weeded zones could be accounted for by the presence of plant material within
sediment samples. Upon immediate visual inspection of sediment samples taken from weeded zones, it was clear that plant material was present within the sediment cores. As these samples were desiccated and pulverized as per the protocol, vegetative matter was assimilated in sample matrix.

5.5 Insects

The lack of aquatic insects within sediment samples coincides with the determined low organic matter content. Lake level fluctuations and wave action are partially responsible for the lack of sufficient sediment and organic matter deposition necessary to support benthic insects. Although the littoral benthos yields no significant insect biomass, milfoil canopies most probably do. Complex interactions between plant, epiphytes and associated grazers undoubtedly exist within milfoil canopies; however they were not measured as part of this study. Large aquatic invertebrates reside primarily on macrophytes (Crowder and Cooper 1982; Morin 1984). As large quantities of predatory fishes thrive in Cave Run Lake (Axon 1981) it must be assumed that they forage successfully within littoral weed beds. The sediments, however, do not appear able to yield sufficient invertebrate biomass to support a large, complex food web.

5.6 Conclusions

The invasion of Cave Run Lake by milfoil has had drastic results. Since it was first observed in the mid-nineties, this invader has successfully exploited nearly the entire lake edge. milfoil appears to be in a "boom cycle" as is characteristic with this species during its introduction (Grace and Wetzel 1978). There are few threats to
its continued establishment, as the only factors limiting it are depth, water level fluctuations, and available light (Titus and Hoover 1991). Although milfoil has established, it did not create the problems found in many other northern lakes. Because the water level fluctuations and bathometry are drastic in this reservoir, the plant will be restricted by water depth. Also, establishment did not displace large stands of native vegetation. Most sites where it has established were previously open water. The shallower zones are still the providence of another invasive species. The question of milfoil’s importance or detriment to this system is debatable however. At present, milfoil has already negatively impacted surface water usage in some areas. This is unfortunate as this impoundment’s primary use is recreation. Although it is possible that its density may increase to some extent it is highly unlikely that milfoil will render Cave Run Lake unable to serve as recreational resource. From a biological standpoint, milfoil has provided littoral flora necessary to the development of both invertebrate and fish species. It provides structure and foraging habitat. The presence of milfoil beds coincides with increased catch per unit effort among anglers (Figure 10). The direct effect of milfoil on fish populations is unclear. To this point, milfoil does not seem to have any deleterious effects on any one fish species, and may be beneficial for production of forage fish. If and to what extent it has positively influenced the environment has yet to be shown. Further studies are necessary to determine if macrophyte beds have higher associated fish biomass than non-weeded zones. As angling alone is a poor estimate of biomass, electrofishing studies would
Muskellunge Angling Success; Cave Run Lake, Kentucky 1991-2000

Figure 10. Number of muskellunge caught on Cave Run Lake, Kentucky; 1991-2000. Reported by the Kentucky Chapter of Muskies Inc. (Grattan et al. 2000)
prove useful in this capacity. Likewise, based on actual fish data, the role of milfoil in the development and behavior of fish species could be better understood. This information will not only aid anglers but may also reveal useful properties of this invasive plant. It would seem that in oligotrophic reservoirs, at least those managed for fishing, any macrophytes are better than none at all. As is characteristic of this plant, it will eventually move into a "bust cycle" where its production will drop off after a plateau. What this phenomenon holds in store for the resident fish population of Cave Run Lake can only be speculated. Macrophytes are undoubtedly good for fish. However this study does not support any advance or decline of any one species based on the presence of milfoil.
6. References Cited


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