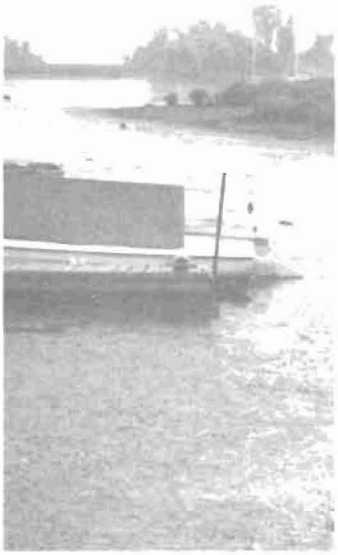
US Army Corps of Engineers



AQUATIC PLANT CONTROL
RESEARCH PROGRAM

MISCELLANEOUS PAPER A-85-5

# MONOECIOUS *HYDRILLA* IN THE POTOMAC RIVER

by

Environmental Laboratory

DEPARTMENT OF THE ARMY
Waterways Experiment Station, Corps of Engineers
PO Box 631, Vicksburg, Mississippi 39180-0631

August 1985
Final Report

Approved For Public Release; Distribution Unlimited

Prepared for US Army Engineer District, Baltimore
Baltimore, Maryland 21203-1715

and DEPARTMENT OF THE ARMY
US Army Corps of Engineers
Washington, DC 20314-1000

Unclassified

SECURITY CLASSIFICATION OF THIS PAGE (When Data Entered)

| REPORT DOCUMENTATION PAGE | | READ INSTRUCTIONS BEFORE COMPLETING FORM |
|---|---|---|
| 1. REPORT NUMBER<br>Miscellaneous Paper A-85-5 | 2. GOVT ACCESSION NO. | 3. RECIPIENT'S CATALOG NUMBER |
| 4. TITLE (and Subtitle)<br>MONOECIOUS *HYDRILLA* IN THE POTOMAC RIVER | | 5. TYPE OF REPORT & PERIOD COVERED<br>Final report |
| | | 6. PERFORMING ORG. REPORT NUMBER |
| 7. AUTHOR(s) | | 8. CONTRACT OR GRANT NUMBER(s) |
| 9. PERFORMING ORGANIZATION NAME AND ADDRESS<br>US Army Engineer Waterways Experiment Station<br>Environmental Laboratory, PO Box 631<br>Vicksburg, Mississippi 39180-0631 | | 10. PROGRAM ELEMENT, PROJECT, TASK AREA & WORK UNIT NUMBERS<br>Aquatic Plant Control Research Program |
| 11. CONTROLLING OFFICE NAME AND ADDRESS<br>US Army Engineer District, Baltimore, Baltimore, Maryland 21203-1715 and DEPARTMENT OF THE ARMY US Army Corps of Engineers, Washington, DC 20314-1000 | | 12. REPORT DATE<br>August 1985 |
| | | 13. NUMBER OF PAGES<br>223 |
| | | 15. SECURITY CLASS. (of this report)<br>Unclassified |
| 14. MONITORING AGENCY NAME & ADDRESS(if different from Controlling Office) | | 15a. DECLASSIFICATION/DOWNGRADING SCHEDULE |

16. DISTRIBUTION STATEMENT (of this Report)

Approved for public release; distribution unlimited.

17. DISTRIBUTION STATEMENT (of the abstract entered in Block 20, if different from Report)

18. SUPPLEMENTARY NOTES

Available from National Technical Information Service, 5285 Port Royal Road, Springfield, Virginia 22161.

19. KEY WORDS (Continue on reverse side if necessary and identify by block number)

*Hydrilla verticillata*
Monoecious *Hydrilla*
Potomac River
Aquatic plants
Aquatic ecology
Aquatic plant management

20. ABSTRACT (Continue on reverse side if necessary and identify by block number)

The report provides:

a. An evaluation of physical and chemical characteristics of the Potomac River that may influence the growth and distribution of *Hydrilla*.

b. A review of literature on the ecology of submersed aquatic vegetation with emphasis on *Hydrilla*.

(Continued)

DD FORM 1 JAN 73 1473 EDITION OF 1 NOV 65 IS OBSOLETE

Unclassified

SECURITY CLASSIFICATION OF THIS PAGE (When Data Entered)

20. ABSTRACT (Continued).

c. A review of the chemical, biological, and mechanical/physical technologies available to control *Hydrilla*.

d. Conclusions and recommendations related to a, b, and c, with specific application to monoecious *Hydrilla* in the Potomac.

# FOREWORD

Funding for the Corps of Engineers' Aquatic Plant Control Program is provided through the Congressional Appropriation, Construction General. One of the requirements for use of Construction General funds is the development of a State Design Memorandum (SDM) and, if deemed necessary, an Environmental Impact Statement (EIS). An SDM is a detailed planning document that evaluates the full range of alternatives for managing a particular aquatic plant problem, and justifies an operational program for its control. Both the SDM and the EIS, while serving separate but related purposes, require that certain statements be made about known conditions, methodologies expected, results, and impacts. Such statements must usually be supported with a more than adequate amount of data and are supplied for information purposes.

When a Corps of Engineers District is in a planning mode to produce the SDM and EIS for the first time, the amount of data and information pertinent to their regional aquatic plant problem is almost always less than adequate. This has been generally true even when the problem plant species is one that has occurred in other regions. When the problem plant is a different biotype, which only recently appeared on the scene, the paucity of needed information and data is even more obvious. Such was the case with the occurrence and spread of the monoecious biotype of hydrilla in the Potomac River. The US Army Engineer District, Baltimore (NAB), was faced with the requirement to develop an SMD and EIS, with a lacking technology base on this type of hydrilla. In addition, data particularly pertinent to this plant as it exists in the Potomac River environment were also lacking. Without an approved SDM and EIS, there could be no authorized operational program for managing the problem.

At the request of NAB, the Corps' Aquatic Plant Control Research Program (APCRP) of the US Army Engineer Waterways Experiment Station (WES), responded with the design of research units and field data collection efforts to fill the technology voids. But, as is the case with natural systems, certain processes to be studied could not be accelerated. Thus, some of the needed information could not be obtained in a timely manner to serve the planning needs of NAB.

As a result of one of the planning meetings between NAB and WES staff members, it was decided that a document could be produced that would "summarize our knowledge" of the more common dioecious hydrilla, along with the not-so-common monoecious biotype. At the same time, inferential conclusions about the responses of the monoecious biotype to various control methods could be drawn from the larger body of knowledge existing on the dioecious biotype. This document is the product of that decision. Only 90 days could be allocated for the effort. Under the circumstances, an extremely outstanding accomplishment was realized.

The APCRP technical experts at WES, their collaborators, and the planning personnel of NAB produced a valued asset that is often overlooked: state of the knowledge in a timely manner. Equally valuable to planners and operations personnel, the document also consolidates a significant amount of information, usually available from many varied sources, in a meaningful context. No doubt,

given the luxury of more time and resources, this document would be much different. The current document, however, may well prove to have additional value other than the original intended purpose. Whether or not a precedent has been established, time will tell. It is a case in point that, provided with the proper impetus, the transfer of technology, in a timely manner, can always be readily accomplished. Those responsible for this effort are to be commended.

J. L. DECELL
Manager, Aquatic Plant
Control Research Program

SUMMARY

*Hydrilla* is a submersed perennial herb in the family Hydrocharitaceae. It has been reported as both dioecious and monoecious, with viable seed produced in the latter. Based on field observations, *Hydrilla* has a geographic range from as far north as 55 deg N latitude in Lithuania and south to North Island in New Zealand at approximately 40 deg S latitude. Although the absolute center of origin is unknown, both Asia and Australia have been suggested. Recent evidence indicates that flowering and propagule production are induced by short days, pointing to origin in temperate regions. However, the majority of established colonies are within and adjacent to the tropics, which indicates a preference for warmer regions.

The dioecious biotype of *Hydrilla* was introduced into Florida, probably from India, in 1958 or 1959. Since that time it has spread north throughout peninsula Florida and westward through the sunbelt states into California. Simultaneously the plant has moved north on the eastern seaboard. The monoecious biotype has been reported to occur in Virginia, Maryland, District of Columbia, Delaware, and North Carolina, where its presence is apparently the result of a separate introduction.

During the past three years, the monoecious biotype of *Hydrilla* has become established in the Potomac River from just north of the Woodrow Wilson Memorial Bridge south to Quantico, Virginia. A survey of this region of the river during the summer of 1984 showed that *Hydrilla* covered nearly 500 acres.* Based on past experience with the dioecious biotype in the southern United States, the continued spread of the monoecious biotype in the Potomac River could present a severe management problem.

## Response to Environmental Conditions

Whereas a good deal of information is available on the ecology of submersed aquatic vegetation in general, most of the reports are based on lacustrine (i.e., lake) studies, and only a portion of the lacustrine studies deal directly with *Hydrilla*. Vegetation in riverine systems has received far less

---

* A table of factors for converting non-SI units of measurement to SI (Metric) units is presented on page xii.

attention, particularly in rivers influenced by estuarine circulation, as in the case in the Potomac River. Assessment of short- or long-term responses of monoecious *Hydrilla* to environmental conditions in the Potomac River is further complicated by an inadequate understanding of its specific environmental tolerances and requirements.

The ability of monoecious *Hydrilla* to become established in the lower reaches of the Potomac River and in the Chesapeake Bay will depend, in part, on its tolerance to salinity. Preliminary evidence indicates an upper tolerance range of 6 to 13 ppt for *Hydrilla*, which is within the range of salinities observed at the mouth of the Potomac River.

The Potomac River is a nutritionally rich environment, thus the spread of monoecious *Hydrilla* is unlikely to be limited by low levels of nutrients. Conversely, the excess nutrients may reduce its distribution by promoting the excessive development of phytoplankton and epiphytic algae, which are highly competitive with *Hydrilla* for available light.

Light is likely to be the most important environmental factor limiting the growth and distribution of monoecious *Hydrilla*. Competition among submersed macrophyte species may be affected by differential abilities to cope with low light conditions. The capacity of monoecious *Hydrilla* to photosynthesize at reduced light is unknown and would be important since this biotype seems to be limited in its ability to extend to full irradiance conditions at the water surface.

Monoecious *Hydrilla* seems to be more tolerant of low temperatures than the dioecious biotype. However, specific temperature requirements for growth and reproduction are unknown. In the southern portions of the United States, the rapid spread of *Hydrilla* is augmented both by its positive response to high water temperatures and by its many efficient modes of vegetative reproduction. The annual regrowth of dioecious *Hydrilla* from subterranean tubers allows this biotype to overwinter, as well as to invade new areas. Equivalent information is not available presently for the monoecious biotype.

The distribution of a variety of aquatic vegetation, including *Hydrilla*, depends on an ability to utilize sediments of widely varying composition. Therefore, to determine the potential distribution of monoecious *Hydrilla* in the Potomac River will probably be dependent on sediment composition: texture and concentrations of reduced substances, nutrients, salinity, and organic constituents.

## Management Strategies

In determining management strategies for control of *Hydrilla*, consideration must be given to the beneficial as well as the negative effects of aquatic vegetation on the environment. By providing direct and indirect sources of food for fish and waterfowl and by giving refuge to a variety of aquatic organisms, moderate densities of submersed vegetation can contribute substantially to habitat enhancement. Aquatic macrophytes are important also as a sink for nutrients, a source of particulate matter, and a factor affecting sedimentation rates, water flow, and water clarity.

A variety of methods are available for control of *Hydrilla*; these can be classified as biological, mechanical/physical, and chemical (Table 1). These methods were reviewed with consideration for their specific applicability, including advantages and disadvantages, in the Potomac River.

### Biological control

The Biological Control Section of the report describes a number of insects that have been identified as potential control agents. Of these, the most promising is a pyralid moth (*Parapoynx diminutalis*). Before the use of this moth as a biocontrol agent for *Hydrilla* control in the Potomac River can be authorized, temperature-tolerance and host-specificity studies will be required.

The grass carp (*Ctenopharyngodon idella*) has been used successfully in a number of areas in the US for control of *Hydrilla*. When carp are introduced in quantities commensurate with the plant problem, native fish, waterfowl, reptiles, and amphibian populations appear to be unaffected. In the Potomac River, released grass carp should remain in the river and not migrate to the Chesapeake Bay, due to their salinity intolerance. A significant disadvantage associated with the use of the grass carp is that they feed indiscriminately and will utilize any submersed vegetation, including desirable species, as a food source.

### Mechanical/physical control

In the Mechanical/Physical Control Section, a number of operational and experimental techniques are described. Among these techniques, the most widely used is mechanical cutting and harvesting. A harvesting test conducted in the Potomac River during the summer of 1984 demonstrated no unique problems associated with harvesting monoecious *Hydrilla*. However, long-term control

Table 1

Comparative Evaluation of Control Techniques for Management of Monoecious *Hydrilla* in the Potomac River

| Control Technique | Areal Treatment | Access Lanes | Duration of Control | Time to Achieve Control | Time of Initial Treatment | Sub-sequent Treat-ment | Effect on other SAV | Disposal Require-ments | Main-tenance Require-ments | Environmental Impacts | Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Grass carp | Yes | No | 2 growing seasons | 1-2 growing season | Early June | Early June | For succu-lent vegeta-tion same as *Hydrilla* | None | None | No direct impacts | May induce blue-green algae growth due to the release of nutrients |
| *Parapoynx diminutalis* | Yes | No | 1 growing season | 2 months | Early June | None | May be non-host specific | None | None | None | Will require additional laboratory & field studies before use is authorized |
| Cutting & harvesting | Yes | Yes | Information not available | Immediate | July* | None | Same as *Hydrilla* | Yes | None | Minimal | Proven effective for short-term control--produces stem fragments |
| Bottom covering materials | No | Yes | Variable-depends on sedimentation rate | Immediate | April | None | Same as *Hydrilla* | None | Annual removal of sediments | Information not available | Should provide at least seasonal control |
| Hydraulic dredging | No | Yes | Depends on depth of dredging | Immediate | Early growing season | None | Same as *Hydrilla* | Yes | None | Turbidity removal of bottom org. | Will result in major mod-ification to river bottom |
| Diver-assisted dredge | No | Yes | Unknown | Immediate | June/July | Unknown | Same as *Hydrilla* | Minimal | None | Slight turbidity | Restricted to small areas & individual plants |
| Mechanical agitation | No | Yes | Unknown | Immediate | May/June | Unknown | Same as *Hydrilla* | Yes | None | Severe | Effectiveness on *Hydrilla* unknown |
| Diquat | Yes | Yes | 6-8 wk | 3-4 wk | Early June or August | August | Same as *Hydrilla* | None | None | None | Only herbicide registered for use in flowing water |
| Chelated copper | Yes | Yes | 6-8 wk | 2-3 wk | Early June or August | August | Same as *Hydrilla* | None | None | Long-term impacts unknown | May result in the long-term presence of copper in the environment |

* Subsequent treatment in September.

of *Hydrilla* by harvesting is unlikely since regrowth of the plant seems to occur rapidly and at about the same density. Considering the cost of intensive long-term harvesting operations, this technique is recommended only for localized areas (e.g., marinas, piers, etc.).

Bottom covering can control the growth of rooted aquatic plants by physically altering the environment. Bottom-covering materials include sand and gravel, sand and gravel laid on an impermeable membrane, impermeable membranes, and various types of permeable fabrics. Among these bottom-covering materials, permeable fabrics and impermeable membranes are generally most effective. Based on recent preliminary studies in the Potomac River, indications are that a bottom-covering technique may provide localized control of *Hydrilla* with limited environmental impacts.

Through dredging, aquatic plants can be removed, and the habitat made unsuitable for further development of aquatic vegetation. However, there are a number of direct short-term environmental impacts associated with dredging. These include turbidity, siltation, and reduced dissolved oxygen levels. For these and other reasons, dredging is recommended as a control technique for localized areas only.

A small-scale hydraulic dredge that is diver operated has been used to control Eurasian watermilfoil in British Columbia. This technique is very slow and labor intensive, but it may be appropriate for limited use in removing small pioneer colonies of spreading plants.

Mechanical agitation of plant-infested sediments has been used experimentally to achieve some control of Eurasian watermilfoil. However, this technique does have adverse water-quality and environmental consequences as well as the probability that it will not be effective in removing the negatively buoyant tubers from the sediment. For these reasons, mechanical agitation is not recommended for control of *Hydrilla*.

Chemical control

Chemical control is the most widely used method in the United States for the control of *Hydrilla*. In the Chemical Control Section, the report provides detailed information on herbicides currently in use; however, for the Potomac River, only copper complexes and diquat are available for use in flowing water.

Copper has been used for many years to control algae and more recently to control *Hydrilla*. Copper concentrations in surficial sediments and in

biological components of aquatic systems can measurably increase following repeated use, possibly resulting in adverse effects; therefore, the use of copper complexes is not recommended as a method for controlling *Hydrilla*.

Among the organic herbicides available, only diquat is registered for use in flowing water. Of the numerous diquat formulations marketed, only "Diquat Water Weed Killer" is registered for control of *Hydrilla*, and it is recommended for use in quiescent or slowly moving water bodies. Under normal use, diquat is unlikely to bioconcentrate significantly or to persist in the tissues of aquatic organisms. Based on past use of diquat for control of *Hydrilla* and on the limited environmental impacts from its use, it is the only herbicide presently recommended for use in the Potomac River.

# PREFACE

This report was prepared for the US Army Engineer District, Baltimore (NAB), for use in the development of a State Design Memorandum and an Environmental Impact Statement regarding the management of monoecious *Hydrilla verticillata* (L.f.) Royle in the Potomac River and its tributaries. Funds were provided by the NAB under appropriation number 96X4902, Revolving Fund, through the Aquatic Plant Control Research Program (APCRP) at the US Army Engineer Waterways Experiment Station (WES), Vicksburg, Miss. Mr. E. Carl Brown of the Office, Chief of Engineers, was APCRP Technical Monitor.

The principal contributors and the individuals primarily responsible for preparation of each chapters are listed at the end of each chapter. Dr. Thomas L. Hart, Chief, Aquatic Processes and Effects Group, Environmental Research and Simulation Division (ERSD), coordinated the preparation of this report. Portions of the report are speculative and were derived from experience gained elsewhere with dioecious *Hydrilla* or other submersed aquatic vegetation.

The work was conducted under the general supervision of Dr. John Harrison, Chief, EL; Mr. Donald L. Robey, Chief, ERSD; Dr. Conrad J. Kirby, Jr., Chief, Environmental Resources Division; and Dr. Lewis E. Link, Jr., Chief, Environmental Systems Division. Mr. J. Lewis Decell was Program Manager of the APCRP at WES.

COL Robert C. Lee was Commander and Director of the WES during the preparation of this report. COL Allen F. Grum, USA, was Director of WES during the publication of this report. Mr. Fred R. Brown and Dr. Robert W. Whalin were Technical Directors.

This report should be cited as follows:

Environmental Laboratory. 1985. "Monoecious *Hydrilla* in the Potomac River," Miscellaneous Paper MP A-85-5, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

TABLE OF CONTENTS

TABLE OF CONTENTS (Continued)

## CONVERSION FACTORS, NON-SI TO SI (METRIC) UNITS OF MEASUREMENT

Non-SI units of measurement used in this report can be converted to SI (metric) units as follows:

| Multiply | By | To Obtain |
|---|---|---|
| acres | 4046.873 | square metres |
| acres per day | 4046.873 | square metres per day |
| acres per hour | 4046.873 | square metres per hour |
| cubic yards | 0.7645549 | cubic metres |
| feet | 0.3048 | metres |
| gallons (US liquid) | 3.785412 | cubic decimetres |
| inches | 25.4 | millimetres |
| miles per hour (US statute) | 1.609347 | kilometres per hour |
| pounds (mass) per acre | 0.000112 | kilograms per square metre |
| square feet | 0.09290304 | square metres |
| tons per acre (mass) | 0.22 | kilograms per square metre |

# CHAPTER I: *HYDRILLA* IN THE POTOMAC ENVIRONMENT

## Overview

The Potomac River estuary is located in the western shore coastal plain of the Chesapeake Bay (Fig. 1). The watershed of this estuary is the second largest of the tributaries of the mid-Atlantic United States (3,799,595 ha), and the river is the second largest tributary of the Chesapeake Bay (646 km long). The average depth of the Potomac River estuary is 5.8 m with a deep channel and adjacent wide, shallow shelf. River flow fluctuates seasonally with the greatest flow recorded during March 1936 (13,707 $m^3$/sec) and the lowest flow during September 1966 (3.4 $m^3$/sec); the average freshwater inflow is 323 $m^3$/sec (51-year average according to the US Geological Survey (USGS) in 1981). The immediate vicinity of the watershed (Washington, D. C., and the surrounding metropolitan area) has a population of about 3 million, which results in huge discharges of sewage with nutrient loads of nitrogen and phosphorus at about 5.4 and 0.45 metric tons per day, respectively (Callender et al. 1984).

## Distribution of *Hydrilla* Biomass

The monoecious biotype of *Hydrilla verticillata* was positively identified in Dyke Marsh, Virginia, in 1982. By 1983, a shoreline survey showed that it was most abundant within 2 to 4 km north and south of Dyke Marsh on the Virginia side of the river about 162 km from the mouth of the Potomac River. A survey of the Potomac River during the summer of 1984 indicated that submersed beds of *Hydrilla* exist from the Woodrow Wilson Bridge (166 km) to Mallows Bay (125 km) (Fig. 2). This area from Quantico, Va. (near Mallows Bay), to Alexandria, Va. (near Woodrow Wilson Bridge), is in the tidal river zone of the Potomac River (Fig. 1) (Callander et al. 1984). A majority of *Hydrilla* surveyed in 1984 occurred 5 km north and south of Dyke Marsh along both the Virginia and Maryland shorelines. The beds varied considerably in density from sparse patches to 100-percent cover (Fig. 2).

The *Hydrilla* biomass, based on dry mass in 0.093-$m^2$ grabs using oyster tongs (Paschal et al. 1982), peaks in early September with substantial mass remaining in October and November (Rybicki et al. 1985). This pattern of

standing crop for *Hydrilla* exhibits peak values later in the summer compared to an earlier peak biomass of species more native to the Chesapeake Bay area, such as *Potamogeton perfoliatus*, *P. pectinatus*, and *Ruppia maritima* (Kemp et al. 1984). Based on 20 samples within *Hydrilla* beds in the Potomac River from 3 July to 8 November 1984, the *Hydrilla* biomass ranged from 20 to 360 g (dry wt)/$m^2$ with a mean (± standard error) of 134.6 (± 18.9) g (dry wt)/$m^2$ (Fig. 3).

An experiment was performed by Rybicki et al. (1985) to determine the recolonization potential of *Hydrilla* removed from 1-$m^2$ plots. At the initiation of the experiment on 3 July 1984, mean biomass was 87 g (dry wt)/$m^2$ (Fig. 4). This mass was removed, and regrowth checked about one month later. The recolonized area reached a peak biomass in early September at nearly twice the original biomass in July. Based on the number of days between sampling, the *Hydrilla* biomass net production varied from 1 to 4 g (dry wt)/$m^2$/day (Fig. 4). No information is available on changes in biomass of other submersed aquatic plants in the Potomac River.

## Sediment Characteristics

The physical (Table 1) and chemical (Table 2) data for sediments in the tidal Potomac River are for areas that are near locations presently inhabited by *Hydrilla*, and although their consistent values along the axis of the river (except for site 4 sampled on 4/81) indicate a homogeneous benthos, they may not be representative of shallow flats colonized by submersed vegetation. A key characteristic among the data for particle size of sediments is that, for most of the stations, sand comprised more than 80 percent of the particles. An exception was at Goose Island where more than 50 percent of the particles were silt sized when sampled in 1981 (Table 1). Only one of these sites, MN-10R (124 km from mouth of the river), was vegetated at the time of sediment sampling; the particle-size distribution at this site was 93:2:5 percent sand: silt:clay.

High sand content is not surprising in the tidal river zone of the Potomac River since tidal currents and low salinity keep materials suspended in the water column. Downstream from this tidal freshwater zone is the transition zone where high sedimentation rates occur. In the downstream zone of the

estuary, the silt and clay particles generally dominate the sediment characteristics. However, environmental conditions in small tributaries and coves adjacent to the tidal freshwater zone where *Hydrilla* presently occurs may also promote the formation of high silt and clay sediments demonstrated at the Goose Island station in 1981 (Table 1).

Nitrogen and phosphorus concentrations of sediment in the tidal Potomac River are summarized in Table 2. Total organic carbon concentrations, listed for three sampling dates, ranged from 1.4 to 16 g/kg (<2 percent dry wt).

## Hydrology and Water Quality

Water-quality and hydrology data were collected by the US Geological Survey (USGS) on the tidal Potomac River at stations designated in Fig. 5 (Blanchard and Coupe 1982). Intervening stations between the two major locations at Alexandria and Quantico were located in areas presently inhabited by *Hydrilla* (see Fig. 2). The water volumes and surface areas of reaches between the sampling stations (Tables 3 and 4) indicate that higher density and areal coverage of *Hydrilla* occurs in the more restricted and lower volumetric areas of the river. The average water depth in this region ranges from 2.2 m near Alexandria to 5.2 m near Hallowing Point.

Tides occur along the entire *Hydrilla* zone of the tidal Potomac River from Quantico to Alexandria (Fig. 6). Tides are semi-diurnal with nearly equal amplitude. Mean tidal amplitude in this *Hydrilla* zone ranges from 0.43 to 0.85 m, and spring tides range from 0.48 to 0.98 m (Blanchard and Coupe 1982). The mean tide levels range from 0.21 to 0.43 m, which may represent about 10 percent of the mean depth of certain areas of the tidal river zone.

Maximum tidal currents in the *Hydrilla* zone of the tidal Potomac River vary from 0.257 to 0.566 m/sec for both flood and ebb tides (Table 5). Minimum current for both tidal periods is 0.0 m/sec. There is no information in various USGS documents that describes the lateral variation in tidal currents, so no estimate of current velocities associated with the shallow littoral zone can be made. Also, current velocities may be influenced by the presence of rooted submersed vegetation in these shallow flat areas of the Potomac River.

Besides the water-quality data collected by USGS at standard sampling stations on the tidal Potomac River (Blanchard and Coupe 1982, Fig. 5),

Paschal et al. (1982) monitored selected water-quality variables in 1978-1980 in the areas surveyed for submersed aquatic plants in 1984 (Fig. 2 and Table 6). From Paschal's survey, water-quality variables were summarized for areas near locations presently inhabited by *Hydrilla* (Table 6). There was generally no salinity recorded for stations more than 150 km from the mouth of the Potomac River during 1979 and 1980, but values from 0.0 to 0.6 parts per thousand (ppt) were measured in 1978 (Table 6). At approximately 123 km from the mouth of the river, salinity ranged from 0.0 to 2.0 ppt in 1980; for other stations, from 1978-1980 most salinity values were equal to or less than 0.5 ppt. The pH, which was only measured in 1978, varied from 6.8 to 9.1 (Table 6).

Data from Blanchard and Coupe (1982) for water year 1981 at the Alexandria and Quantico stations were used to determine the seasonal nature of certain water-quality variables (Fig. 7). Conductivities increased during late fall and early winter, with peak concentrations of salts occurring during January and February. February was also the month with the greatest range of conductivity since this period was the beginning of the freshet that decreased conductivities to less than 200 μmhos. Minimum values of conductivity occurred throughout the spring and summer (Fig. 7), which corresponds to the growing season of *Hydrilla*. Conductivity values at Quantico were about an order of magnitude greater than at Alexandria. The maximum conductivity during this survey was 11,490 μmhos at Quantico during January, which corresponded to a salinity of about 7 ppt.

The pH ranged from 6.4 to 8.9 for both stations during water year 1981, and no distinct seasonal pattern was observed at either station. Values were generally higher at the Quantico station compared to Alexandria apparently because of the greater buffer capacity of estuarine waters at the downstream station.

Total suspended solid (TSS) concentrations (comprised of organic and inorganic particles) were generally higher in later winter and early spring at both stations; this pattern was obviously related to increased freshwater discharge during these months. From February to April, peak TSS concentrations were 100 to 180 mg/ℓ at Alexandria compared to 80 to 90 mg/ℓ at Quantico (Fig. 7).

Limited information on the size distribution of TSS at Alexandria and Quantico during April 1981 (Blanchard and Coupe 1982) suggests that 51 to

77 percent of the particles suspended in the water were clay (≤0.004 mm) and that the remaining particles were silt with 3 percent or less sand (Table 7). The distribution in particle sizes was similar between the two stations and no relationship was observed between distribution and TSS concentration. The TSS concentrations at both stations were 50 mg/ℓ or less during the growing season of *Hydrilla* from May to October 1981. The dominant particle fraction at this time of year is expected to be of organic origin since river flow is also near its minimum.

The amount of light in the water column at stations in the channel was based on Secchi disk depths, which represent the depth of 1 percent light penetration (Blanchard and Coupe 1982). At Alexandria, Secchi disk depths ranged from 0.25 to 0.91 m during the growing season (May thru September). Secchi disk depths were slightly lower at Quantico with a range of 0.25 to 0.63 m. Water transparency was greater at both stations during the winter with Secchi disk depths of 1.82 and 1.67 m at Alexandria and Quantico, respectively. Based on data from Paschal et al. (1982) for stations nearer areas vegetated by submersed grasses (but near the river channel), Secchi disk depths in the shallow littoral zone were similar to values for the main stem channel with most Secchi depths less than 0.5 m during the summer.

Dissolved oxygen concentrations ranged from 4.4 to 14.2 mg/ℓ for both Alexandria and Quantico, and concentrations were generally higher in the winter months (Fig. 7). This zone of Potomac River is apparently not affected by anoxia that is a common water quality problem in other areas of the Chesapeake Bay (Officer et al. 1984).

There are very few Potomac River measurements of dissolved organic carbon and alkalinity reported in the literature, but most of these numbers are for months during the growing season of *Hydrilla*. Dissolved organic carbon ranged from 0.8 to 6.0 mg/ℓ in the river channel at Alexandria compared to lower values from 0.3 to 4.0 at Quantico. These measurements were made during July and August. Alkalinity values ranged from 48 to 84 mg/ℓ at both stations from July to September. Values were slightly less at Quantico.

## Contributors


| | |
|---|---|
| Dr. David C. Beckett | US Army Engineer Waterways Experiment Station<br>ATTN: WESER-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. Kurt D. Getsinger | US Army Engineer Waterways Experiment Station<br>ATTN: WESES-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. Donald Graetz | Soil Science Dept.<br>106 Newell Hall<br>University of Florida<br>Gainesville, FL 32611 |
| Dr. Jerry F. Hall | US Army Engineer Waterways Experiment Station<br>ATTN: WESES-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. John Rodgers, Jr. | Institute of Applied Sciences<br>North Texas State University<br>Denton, TX |
| Dr. Harish Sikka | Director, Great Lakes Research Center<br>Suny Lab<br>1300 Allenwood Ave<br>Buffalo, NY 14222 |
| Dr. Robert Twilley | Horn Point Laboratories<br>PO Box 775<br>Cambridge, MD 21613 |
| Dr. Thai K. Van | USDA/ARS<br>3205 SW College Avenue<br>Ft. Lauderdale, FL 33314 |
| Dr. Howard E. Westerdahl* | US Army Engineer Waterways Experiment Station<br>ATTN: WESES-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |


---

* Dr. Westerdahl was primarily responsible for the development of this chapter and should be used as a point of contact regarding its content.

## References


Blanchard, S. F., and R. H. Coupe, Jr. 1982. Water quality of the tidal Potomac River and estuary and hydrological data report, 1981 water year. USGS Open-File Report 82-575. Washington, D.C.

Callender, E., V. Carter, D. C. Hahl, K. Hitt, and B. I. Schultz. 1984. A water-quality study of the tidal Potomac River and estuary -- An overview. USGS Water-Supply Paper 2233. Washington, D.C.

Kemp, W. M., W. R. Boynton, J. C. Stevenson, R. R. Twilley, and L. G. Ward. 1984 (In Press). Influences of submersed vascular plants on ecological processes in upper Chesapeake Bay, pp. 000-000. In V. Kennedy (ed.), The estuary as a filter.

Officer, C. B., R. B. Biggs, J. L. Taft, L. E. Cronin, M. A. Tyler, and W. R. Boynton. 1984. Chesapeake Bay anoxia: origin development and significance. Science, 223:51-53.

Paschal, J. E., Jr., D. R. Miller, N. C. Bartow, and V. Carter. 1982. Submersed aquatic vegetation in the tidal Potomac River and estuary of Maryland, Virginia, and the District of Columbia. Hydrologic data Report, May, 1978 to November, 1981. USGS Open-File Report 82-694. Washington, D.C.

Rybicki, N. B., V. Carter, R. T. Anderson, and T. J. Trombley. 1985 (In Press). *Hydrilla verticillata* in the tidal Potomac River, 1983 and 1984. USGS Open-file Report.

Table 1

Particle-size Distribution of Sediments in the Tidal Potomac River

Near Areas Inhabited by *Hydrilla* in July 1984*

| Site Description | Site Number | River Distance km | Sampling Date | Vegetated | Particle Type - %** | | | |
|---|---|---|---|---|---|---|---|---|
| | | | | | Gravel | Sand | Silt | Clay |
| Goose Island (G1-1R) | 4 | 172 | 5/80 | No | - | 42.2 | 57.8 | - |
| | | | 4/81 | No | 0 | 14.7 | 66.8 | 18.5 |
| Rosier Bluff (PY-1R) | 5 | 166 | 4/81 | No | 0 | 93.9 | 3.8 | 2.3 |
| Elodea Cove (PY-8R) | 6 | 154 | 5/80 | No | - | 96.2 | 3.8 | - |
| | | | 4/81 | No | 0 | 80.8 | 14.3 | 5.0 |
| MN-10R | 9 | 124 | 8/78 | Yes | 0 | 92.6 | 1.86 | 5.54 |

* From Paschal et al. 1982

** Size of particle types as follows:

Gravel = > 2 mm

Sand = < 2 and > 0.062 mm

Silt = < 0.062 and > 0.004 mm

Clay = < 0.004 mm

Table 2

Nutrient Concentration of Bottom Sediments in the Tidal Potomac River Near Areas Inhabited by *Hydrilla* in July 1984*

| Site Description | Site Number | River Distance (km) | Sampling Date | Vegetated | Sediment Nutrients** TN | $NH_4$ | DIP | TP | TC |
|---|---|---|---|---|---|---|---|---|---|
| Goose Island | 4 | 172 | 5/79 | No | - | - | - | - | 16 |
| (G1-1R) | | | 7/80 | No | 2,400 | - | - | 440 | |
| | | | 4/81 | No | 3,300 | 34 | 470 | - | |
| | | | 4/81 | No | 802 | 31 | 260 | - | |
| | | | 4/81 | No | 2,300 | 45 | 380 | - | |
| Rosier Bluff | 5 | 166 | 5/79 | No | - | - | - | - | 1.4 |
| (PY-1R) | | | 4/81 | No | 734 | 11 | 130 | - | |
| | | | 4/81 | No | 443 | 4 | 110 | - | |
| | | | 4/81 | No | 602 | 6 | 140 | - | |
| Elodea Cove | 6 | 154 | 5/80 | No | - | - | - | - | 4.3 |
| (PY-8R) | | | 7/80 | No | 11,100 | - | - | 260 | |
| | | | 8/80 | No | 760 | - | - | 220 | |
| | | | 4/81 | No | 1,100 | 22 | 290 | - | |
| | | | 4/81 | No | 1,400 | 26 | 340 | - | |
| | | | 4/81 | No | 662 | 25 | 180 | - | |

* From Paschal et al. 1982

** TN = total nitrogen; $NH_4$ = ammonium; DIP = inorganic phosphorus as P; TP = Total phosphorus; TC = total carbon.
Total carbon is given in grams per kilogram of sediment; all other data are given in milligrams per kilogram of sediment.

Table 3

Volumes and Surface Areas for Selected Reaches of Tidal Potomac River and Tributaries

| Mid-reach Name | Location in km (nmi) from Mouth of Potomac River | | Average Depth m | Mean Low-Water Surface Area $10^6$ $m^2$ | Mean Low + 1/2 Inter-tidal Volume $10^6$ $m^3$ | Accumulated Volume $10^6$ $m^3$ | Accumulated Surface Area, $10^6$ $m^2$ |
|---|---|---|---|---|---|---|---|
| | From | To | | | | | |
| Memorial Bridge | 181.6 (98) | 176.0 (95) | 3.0 | 3.08 | 9.1 | 9.1 | 3.1 |
| Giesboro Point | 176.0 (95) | 172.3 (93) | 4.6 | 7.18 | 33.1 | 42.2 | 10.3 |
| Marbury Point | 172.3 (93) | 169.6 (91.5) | 2.9 | 3.76 | 11.1 | 53.3 | 14.1 |
| Alexandria | 169.6 (91.5) | 166.8 (90) | 2.2 | 3.98 | 8.8* | 62.1 | 18.1 |
| Rosier Bluff | 166.8 (90) | 163.1 (88) | 2.8 | 7.74 | 21.3 | 83.4 | 25.8 |
| Hatton Point | 163.1 (88) | 155.7 (84) | 3.6 | 10.33 | 37.0 | 120.4 | 36.1 |
| Marshall Hall | 155.7 (84) | 148.3 (80) | 2.8 | 18.94 | 53.5 | 173.9 | 55.0 |
| Hallowing Point | 148.3 (80) | 140.8 (76) | 5.2 | 12.89 | 67.0 | 240.9 | 67.9 |
| Indian Head | 140.8 (76) | 132.6 (71) | 3.5 | 48.13 | 167.4 | 408.3 | 116.0 |
| Quantico | 132.6 (71) | 124.2 (67) | 3.4 | 49.18 | 168.8 | 577.1 | 165.2 |

* 1.9 for Maryland channel

6.9 for Virginia channel

Table 4

Volumes and Surface Areas for Selected Reaches of Tidal Potomac River with Tributaries Excluded

| Mid-reach Name | Location km (nmi) from Mouth of Potomac River: From | To | Average Depth m | Mean Low-Water Surface Area $10^6$ $m^2$ | Mean Low + 1/2 Inter-tidal Volume $10^6$ $m^3$ | Accumulated Volume $10^6$ $m^3$ | Accumulated Surface Area, $10^6$ $m^2$ |
|---|---|---|---|---|---|---|---|
| Memorial Bridge | 181.6 (98) | 176.0 (95) | 3.0 | 3.1 | 9.1 | 9.1 | 3.1 |
| Giesboro Point | 176.0 (95) | 172.3 (93) | 4.4 | 2.6 | 11.3 | 20.4 | 5.7 |
| Marbury Point | 172.3 (93) | 169.6 (91.5) | 2.9 | 3.8 | 11.1 | 31.5 | 9.5 |
| Alexandria | 169.6 (91.5) | 166.8 (90) | 2.2 | 3.3 | 7.2* | 38.7 | 12.8 |
| Rosier Bluff | 166.8 (90) | 163.1 (88) | 3.3 | 5.5 | 18.3 | 57.0 | 18.3 |
| Hatton Point | 163.1 (88) | 155.7 (84) | 3.9 | 8.8 | 34.8 | 91.8 | 27.1 |
| Marshall Hall | 155.7 (84) | 148.3 (80) | 2.8 | 15.9 | 45.3 | 137.1 | 43.0 |
| Hallowing Point | 148.3 (80) | 140.8 (76) | 5.6 | 11.7 | 65.1 | 202.2 | 54.7 |
| Indian Head | 140.8 (76) | 132.6 (71) | 4.9 | 25.9 | 126.6 | 328.8 | 80.6 |
| Quantico | 132.6 (71) | 124.2 (67) | 5.7 | 23.1 | 132.4 | 461.2 | 103.7 |

* 1.5 for Maryland channel

5.7 for Virginia channel

Table 5

Current Differences and Other Constants

| No. | Place | Meter Depth ft | Position Lat. ° N ' | Position Long. ° W ' | Time Differences Min. before Flood h.m. | Time Differences Flood h.m. | Time Differences Min. Before Ebb h.m. | Time Differences Ebb h.m. | Speed Ratios Flood | Speed Ratios Ebb | Average Speeds and Directions * Minimum Before Flood Knots (deg.) | Average Speeds and Directions * Maximum Flood Knots (deg.) | Average Speeds and Directions * Minimum Before Ebb Knots (deg.) | Average Speeds and Directions * Maximum Ebb Knots (deg.) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | POTOMAC RIVER | | | | | | | | | | | | | |
| | Cornfield Point | | | | | | | | | | | | | |
| 4020 | 1 mile south of---------- | | 38 02 | 76 21 | Current irregular | | | | | | 0.0 - - | 0.5 (310) | 0.0 - - | 0.5 (130) |
| 4025 | midchannel---------- | | 38 01.1 | 76 21.3 | +4 00 | +4 00 | +4 00 | +4 00 | 0.5 | 0.4 | 0.0 - - | 0.5 (280) | 0.0 - - | 0.6 (110) |
| 4030 | 3.8 miles south of--------- | | 37 59.4 | 76 21.5 | +3 45 | +3 45 | +3 45 | +3 45 | 0.7 | 0.4 | 0.0 - - | 0.7 (315) | 0.0 - - | 0.6 (100) |
| 4035 | Fort Point, St. Marys River--- | | 38 07.8 | 76 26.9 | Current weak and variable | | | | | | | | | |
| 4040 | Yeocomico River entrance------ | | 38 02.1 | 76 31.2 | Current weak and variable | | | | | | | | | |
| | Piney Point | | | | | | | | | | | | | |
| 4045 | 0.2 mile south of---------- | | 38 07.8 | 76 32.0 | +3 00 | +3 00 | +3 00 | +3 00 | 1.3 | 0.7 | 0.0 - - | 1.3 (280) | 0.0 - - | 0.6 (146) |
| 4050 | midchannel---------- | | 38 06.9 | 76 32.5 | +3 48 | +3 40 | +3 43 | +3 51 | 0.4 | 0.4 | 0.0 - - | 0.4 (290) | 0.0 - - | 0.6 (160) |
| 4055 | 2.2 miles south of--------- | | 38 05.9 | 76 33.1 | +3 00 | +3 00 | +3 00 | +3 00 | 0.5 | 0.3 | 0.0 - - | 0.5 (280) | 0.0 - - | 0.5 (130) |
| 4060 | Lower Machodoc Creek entrance- | | 38 08.7 | 76 39.3 | Current weak and variable | | | | | | | | | |
| 4065 | White Point, Nomini Creek entrance---------- | | 38 08.1 | 76 43.3 | +3 35 | +3 35 | +3 35 | +3 35 | 1.2 | 0.8 | 0.0 - - | 1.2 (155) | 0.0 - - | 1.2 (335) |
| 4070 | Breton Bay entrance---------- | | 38 14.5 | 76 41.7 | +2 20 | +2 20 | +2 20 | +2 20 | 0.6 | 0.3 | 0.0 - - | 0.6 (030) | 0.0 - - | 0.4 (200) |
| 4075 | St. Clements Bay entrance----- | | 38 14.5 | 76 43.7 | Current weak and variable | | | | | | | | | |
| 4080 | St. Clements I., 1.8 miles southeast of---------- | | 38 11.7 | 76 42.5 | +4 45 | +4 45 | +4 45 | +4 45 | 0.4 | 0.6 | 0.0 - - | 0.4 (250) | 0.0 - - | 0.9 (085) |
| 4085 | St. Clements I., 1.1 miles southwest of---------- | | 38 11.57 | 76 45.67 | +4 31 | +4 54 | +4 44 | +4 34 | 0.6 | 0.5 | 0.0 - - | 0.6 (281) | 0.0 - - | 0.8 (099) |
| 4090 | Rock Point, Wicomico River entrance---------- | | 38 16.4 | 76 49.3 | +3.09 | +3 41 | +3 53 | +3 22 | 0.5 | 0.4 | 0.0 - - | 0.5 (019) | 0.0 - - | 0.6 (174) |
| 4095 | Swan Point---------- | | 38 16.4 | 76 56.7 | +6 25 | +6 25 | +6 25 | +6 25 | 0.3 | 0.5 | 0.0 - - | 0.3 (350) | 0.0 - - | 0.8 (140) |
| 4100 | Dahlgren Harbor Channel------- | | 38 18.90 | 77 01.93 | Current weak and variable | | | | | | | | | |
| 4105 | Upper Machodoc Creek entrance- | | 38 19 | 77 02 | Current irregular | | | | | | 0.0 - - | 0.3 (270) | 0.0 - - | 0.3 (090) |
| 4110 | Persimmon Point---------- | | 38 22.1 | 76 59.4 | +7 10 | +7 10 | +7 10 | +7 10 | 1.2 | 0.9 | 0.0 - - | 1.2 (335) | 0.0 - - | 1.4 (175) |
| 4115 | Potomac River Bridge, 0.4 mile south of---------- | | 38 21.38 | 76 59.20 | +6 54 | +7 01 | +7 19 | +7 17 | 0.9 | 0.9 | 0.0 - - | 0.9 (000) | 0.0 - - | 1.4 (165) |
| 4120 | Chapel Point, Port Tobacco River---------- | | 38 27.9 | 77 02.2 | Current weak and variable | | | | | | | | | |
| 4125 | Maryland Point---------- | | 38 20.8 | 77 11.8 | +7 15 | +7 15 | +7 15 | +7 15 | 1.1 | 0.9 | 0.0 - - | 1.1 (270) | 0.0 - - | 1.4 (080) |
| 4130 | Quantico---------- | | 38 31.3 | 77 16.6 | +7 25 | +7 25 | +7 25 | +7 25 | 0.7 | 0.6 | 0.0 - - | 0.7 (020) | 0.0 - - | 0.9 (200) |
| 4135 | Quantico Creek entrance------- | | 38 31.7 | 77 17.3 | +7 00 | +7 00 | +7 00 | +7 00 | 0.5 | 0.3 | 0.0 - - | 0.5 (305) | 0.0 - - | 0.5 (115) |
| 4140 | Freestone Point, 2.3 miles east of---------- | | 38 35.78 | 77 11.88 | +8 16 | +8 28 | +8 29 | +8 28 | 0.7 | 0.5 | 0.0 - - | 0.7 (030) | 0.0 - - | 0.7 (229) |
| 4145 | Hallowing Point---------- | | 38 38.70 | 77 07.65 | +8 31 | +8 24 | +8 33 | +8 19 | 1.1 | 0.7 | 0.0 - - | 1.1 (345) | 0.0 - - | 1.1 (149) |
| 4150 | Jones Point, Alexandria------- | | 38 47.62 | 77 02.23 | +8 55 | +8 30 | +9 06 | +8 41 | 1.0 | 0.6 | 0.0 - - | 1.0 (352) | 0.0 - - | 0.9 (171) |
| 4155 | Hains Point---------- | | 38 51.08 | 77 01.32 | +8 39 | +9 00 | +9 01 | +8 16 | 0.6 | 0.2 | 0.0 - - | 0.6 (359) | 0.0 - - | 0.3 (176) |
| 4160 | Anacostia River entrance------ | | 38 51.8 | 77 00.6 | Current weak and variable | | | | | | | | | |
| 4165 | South Capitol Street Bridge--- | | 38 52.07 | 77 00.38 | Current weak and variable | | | | | | | | | |
| 4170 | Washington Channel, Washington, D.C---------- | | 38 51.8 | 77 01.2 | Current weak and variable | | | | | | | | | |
| 4175 | Virginia Channel, Washington, D.C. <13>---------- | | 38 52 | 77 02 | - - - - - - - - - - - - | | | | - - - - | | - - - - | - - - - | - - - - | 0.6 (145) |

* (Knots × 0.5148) = M/S current velocity

Table 6

Water-quality Data for the Tidal Potomac River at Stations in the Area Inhabited by *Hydrilla** in July 1984

| | River | 1978** | | | | | 1979** | | | | | 1980** | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Transect | Distance km | Date | pH | SAL °/oo | COND µmhos | TEMP °C | Date | pH | SAL °/oo | COND µmhos | TEMP °C | Date | pH | SAL °/oo | COND µmhos | TEMP °C |
| PY-01R | 166 | 5/25 | 6.8 | | 200 | 21.5 | 5/30 | - | 0.5 | 700 | 18.5 | 6/06 | - | 0.0 | 180 | 23.0 |
| | | 8/01 | 7.3 | 0.6 | 900 | 27.9 | | | | | | 6/26 | - | 0.0 | 145 | 25.0 |
| | | | | | | | | | | | | 7/10 | - | 0.0 | 305 | - |
| | | | | | | | | | | | | 8/09 | - | 0.0 | 340 | 32.0 |
| | | | | | | | | | | | | 8/13 | - | 0.0 | 350 | - |
| | | | | | | | | | | | | 9/12 | - | 0.0 | 340 | 26.0 |
| PY-03R | 162 | 8/01 | - | 0.6 | 800 | 26.9 | | | | | | | | | | |
| PY-05R | 158.7 | 5/25 | 6.9 | - | 180 | 21.9 | 9/11 | - | 0.0 | 115 | 23.0 | | | | | |
| | | 8/01 | 7.6 | 0.0 | 325 | - | | | | | | | | | | |
| PY-06R | 157.9 | 5/26 | 7.5 | - | 170 | 20.6 | | | | | | 6/06 | - | 0.0 | 205 | 23.0 |
| | | 8/01 | - | 0.5 | 900 | 27.4 | | | | | | | | | | |
| PY-08R | 154.2 | 8/01 | - | 0.5 | 900 | 27.7 | | | | | | 7/07 | - | 0.0 | 220 | 20.0 |
| | | | | | | | | | | | | 7/28 | - | 0.0 | 450 | 26.5 |
| | | | | | | | | | | | | 10/14 | - | 0.0 | 320 | 15.0 |
| PY-OT | | 5/25 | 8.6 | - | 150 | 21.8 | 9/11 | - | 0.0 | 162 | 23.0 | 5/22 | - | 0.0 | 210 | 24.0 |
| | | 8/01 | 6.8 | 0.0 | 170 | 22.5 | | | | | | 6/06 | - | 0.0 | 187 | 23.0 |
| | | | | | | | | | | | | 6/26 | - | 0.0 | 120 | 26.0 |
| | | | | | | | | | | | | 8/09 | - | 0.0 | 300 | 31.5 |
| | | | | | | | | | | | | 9/12 | - | 0.0 | 240 | 26.0 |
| MN-10R | 124.2 | 5/31 | 7.1 | - | 160 | 22.3 | 5/31 | - | 0.5 | 700 | 21.4 | 6/30 | - | 0.0 | 185 | 25.0 |
| | | 8/02 | 9.1 | 0.0 | 280 | 28.0 | 9/24 | - | 0.0 | 122 | 19.8 | | | | | |
| | | 10/12 | 8.7 | 0.3 | 900 | 20.0 | | | | | | | | | | |
| MP-01R | 122.6 | | | | | | | | | | | 6/30 | - | 0.0 | 185 | 23.5 |
| | | | | | | | | | | | | 8/23 | - | 2.0 | 3200 | - |

* From Paschal et al. (1982).

** Column headings are defined as follows:

pH = acidity measurement
SAL = salinity
COND = conductivity
TEMP = water temperature.

Table 7

Concentration and Size Distribution of Total Suspended Sediment in the Potomac River in 1981*

| Date | Time | Concentration TSS (mg/ℓ) | Particle Size, mm - Percent Passing** | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | <.500 | <.250 | <.125 | <.062 | <.031 | <.016 | <.008 | <.004 | <.002 |
| | | | | | | Alexandria, Va. (168 km) | | | | | |
| April 15 | 1200 | 135 | - | 100 | 100 | 100 | 93 | 84 | 69 | 51 | 32 |
| | 1210 | 125 | - | 100 | 99 | 99 | 97 | 88 | 80 | 57 | 41 |
| | 1300 | 51 | - | 100 | 100 | 99 | 95 | 88 | 75 | 61 | 45 |
| April 16 | 1140 | 105 | - | 100 | 100 | 100 | 99 | 97 | 92 | 77 | 58 |
| | 1335 | 149 | - | 100 | 100 | 100 | 98 | 93 | 87 | 63 | 43 |
| April 17 | 1130 | 110 | - | - | 100 | 100 | 97 | 91 | 76 | 60 | 37 |
| | 1145 | 86 | 100 | 99 | 99 | 99 | 98 | 94 | 80 | 59 | 36 |
| | 1200 | 48 | - | - | - | 100 | 97 | 95 | 90 | 75 | 52 |
| | 1215 | 58 | 100 | 99 | 99 | 98 | 97 | 94 | 89 | 76 | 60 |
| | | | | | | Quantico, Va. (126 km) | | | | | |
| April 16 | 0920 | 62 | 100 | 99 | 99 | 99 | 99 | 95 | 90 | 72 | 48 |
| April 17 | 0920 | 59 | 99 | 99 | 99 | 97 | 95 | 89 | 80 | 59 | 36 |
| | 0930 | 67 | | | | | | 96 | 82 | 63 | 40 |

* From Blanchard and Coupe (1982)

** Sand, silt, and clay particle sizes as follows:
Sand = < 2 and > 0.062 mm
Silt = < 0.062 and > 0.004 mm
Clay = < 0.004 mm

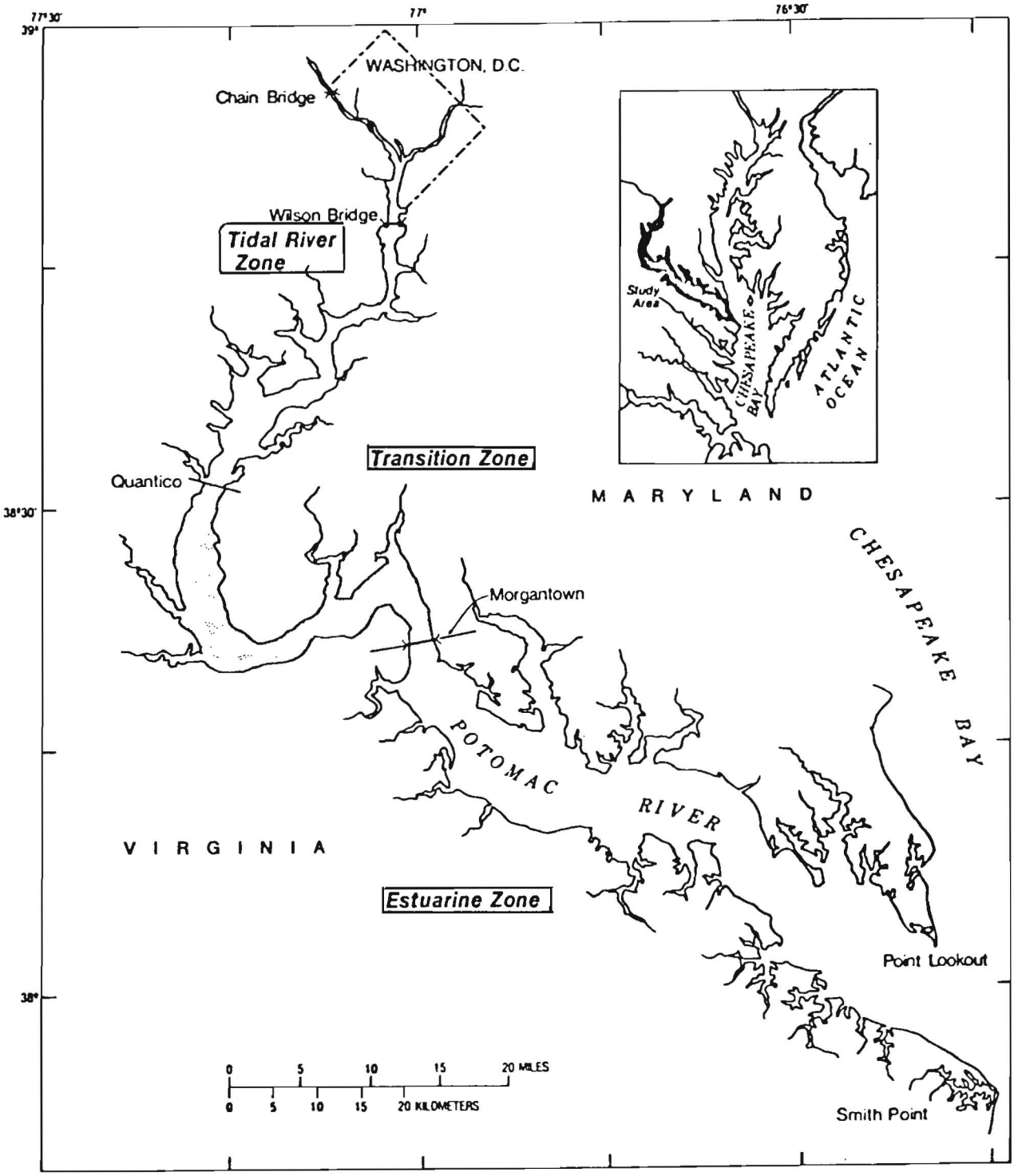



Figure 1. Tidal Potomac River and estuary

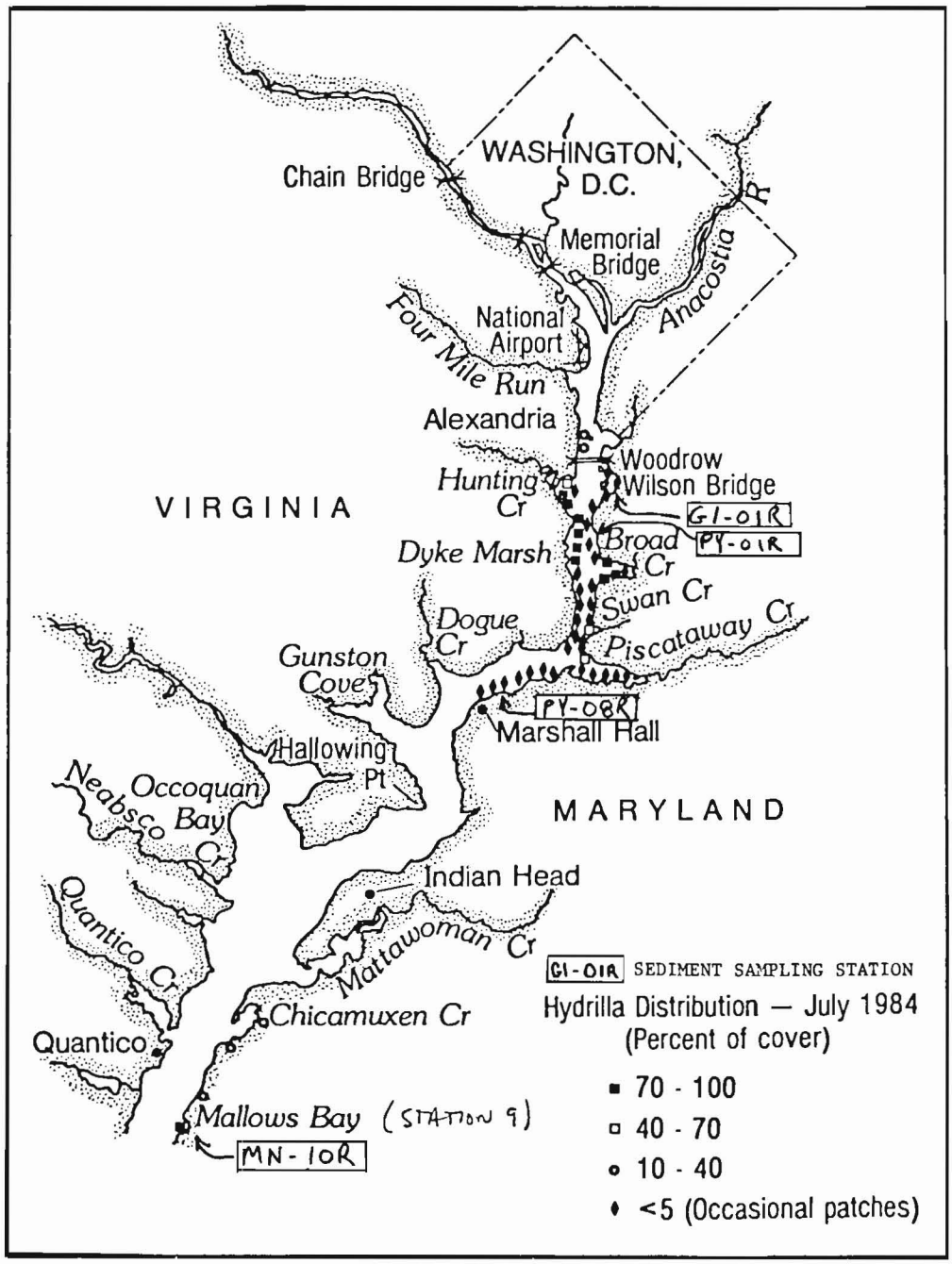



Figure 2. Vicinity map showing sediment sampling stations and *Hydrilla* distribution in 1984 (Rybicki et al. 1985)

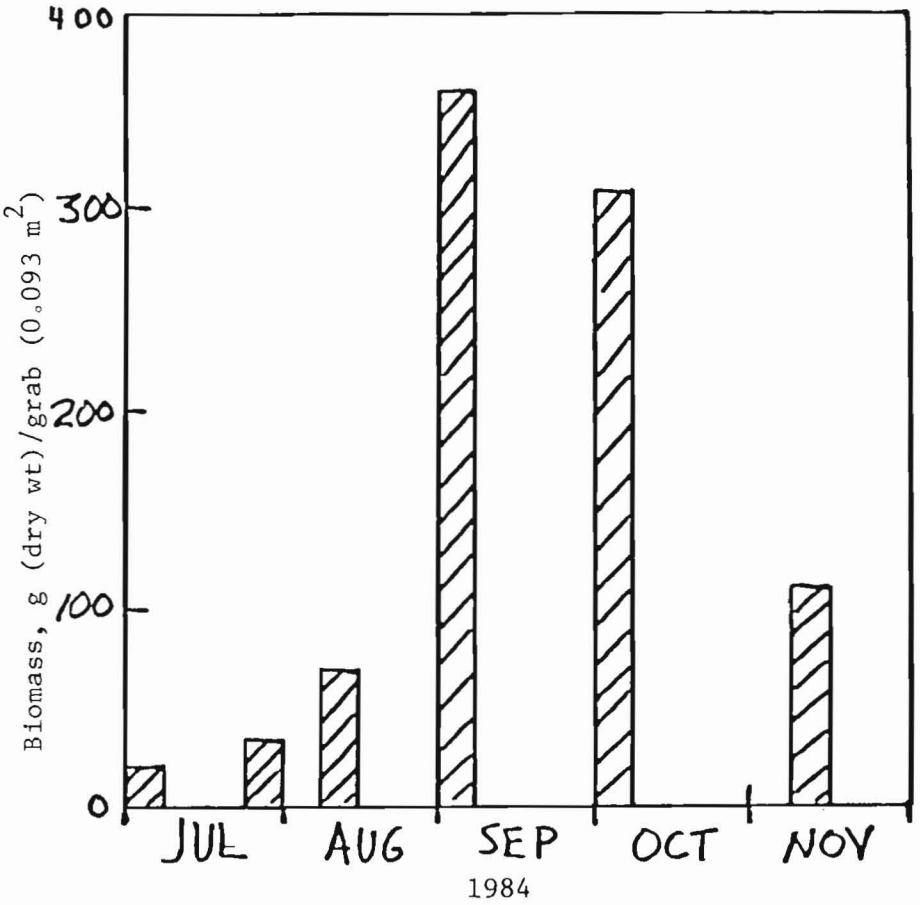



Figure 3. Biomass of *Hydrilla* in the Potomac River sampled 3 July to 8 November 1984 (Rybicki et al. 1985)

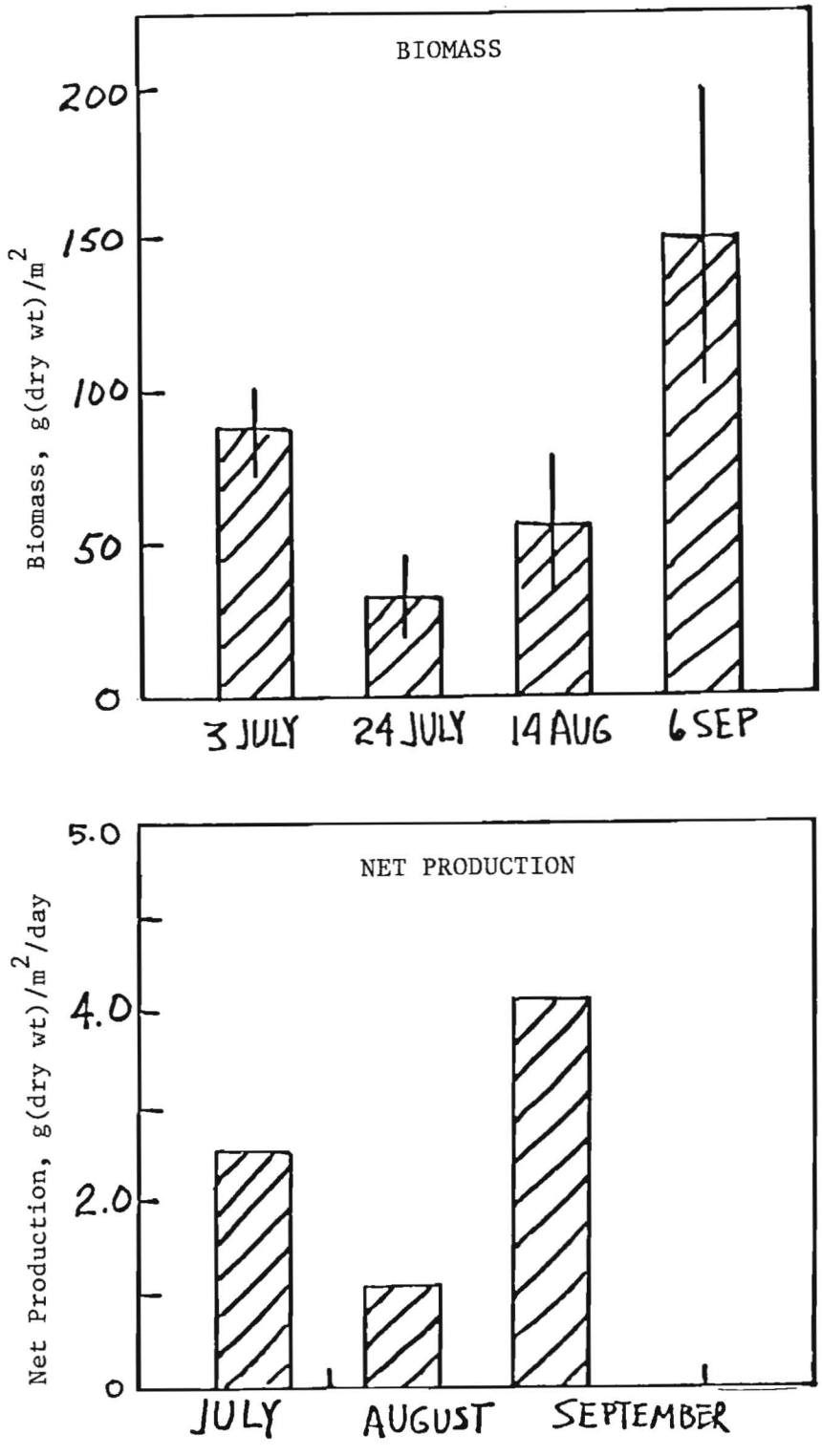



Figure 4. *Hydrilla* biomass and net production in the Potomac River, 3 July to 6 September 1984 (Rybicki et al. 1985)

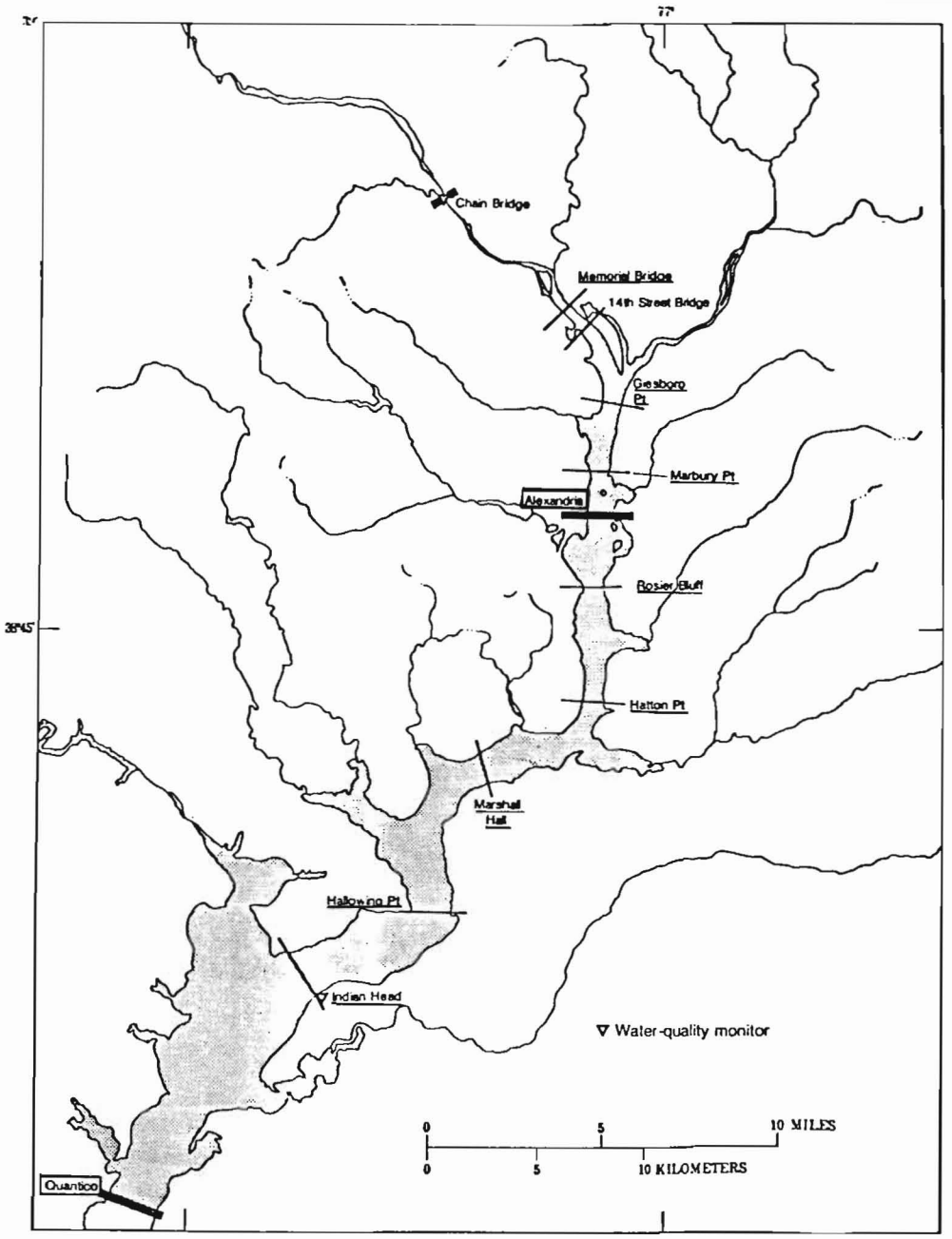



Figure 5. Tidal river zone showing major hydrology and water-quality sampling stations (wide lines), intervening sampling stations (narrow lines), and mid-reach names

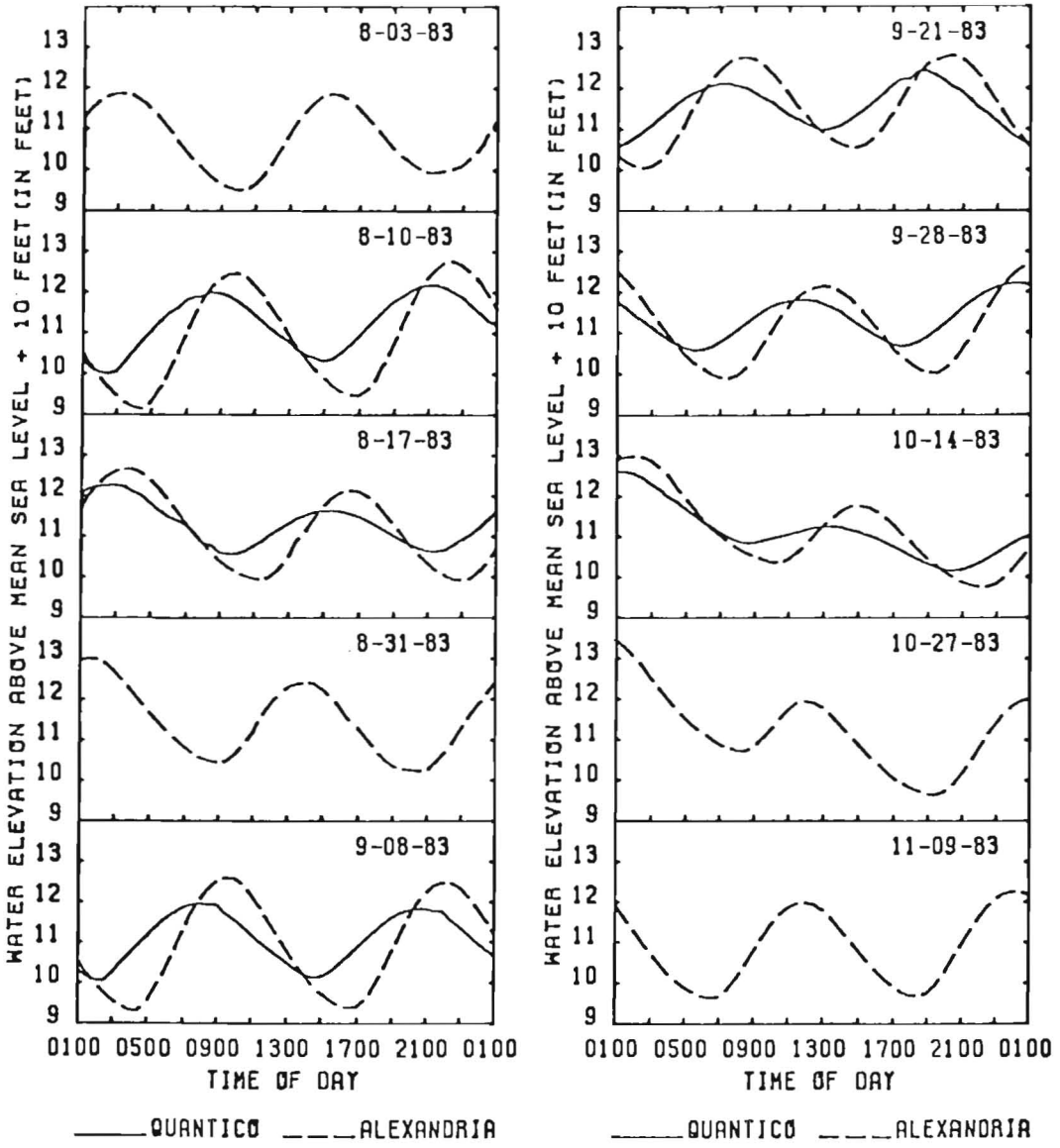



Figure 6. Tide stage measured at Alexandria, Va., and Quantico, Va., for water-quality sampling days

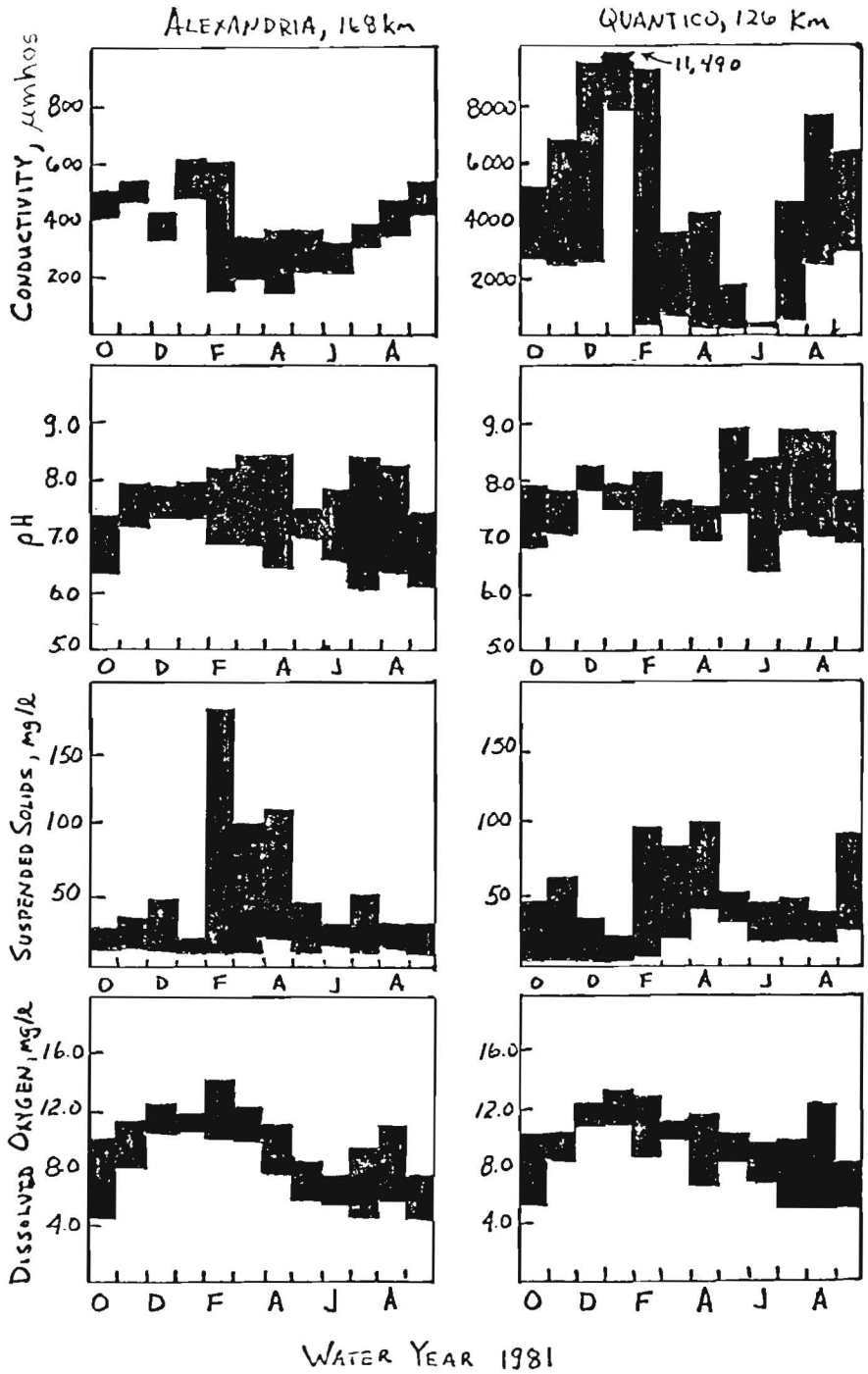



Figure 7. Seasonal nature of certain water-quality variables

# CHAPTER II: ECOLOGY

## Overview

*Hydrilla* is a submersed perennial herb in the family Hydrocharitaceae. It has a wide geographic range, having been reported as far as 55 deg N latitude in Lithuania and as far south as North Island in New Zealand, approximately 40 deg S latitude. While the absolute center of origin is unknown, both Asia and Australia have been suggested (Cook and Luond 1982). Recent evidence indicates that flowering and propagule production are induced by short days, pointing to origin in temperate regions. However, the majority of established colonies are within and adjacent to the tropics, indicating a preference for warmer regions.

*Hydrilla* has been reported as both dioecious and monoecious with viable seed produced in the monoecious biotype. The biology, world distribution, and taxonomy of *Hydrilla* have been comprehensively reviewed by Cook and Luond (1982), Pieterse (1981), and Swarbrick et al. (1982). The morphology of *Hydrilla* has been thoroughly examined by Yeo et al. (1984). These works should be consulted for more comprehensive treatment of the above subject matters.

The dioecious biotype of *Hydrilla* was introduced into Florida, probably from India, in 1958 or 1959 (Blackburn et al. 1969). Since that time it has spread throughout Florida and westward through the sunbelt states into California. Simultaneously the plant has moved up the eastern seaboard (Steward et al. 1984).

The monoecious biotype has been reported in Virginia, Maryland, District of Columbia, Delaware, and North Carolina; its presence is apparently the result of a separate introduction to this country, although the foreign source has not been identified (Steward et al. 1984). The monoecious biotype may also occur in Pennsylvania, since it has been reported in the Susquehanna Flats of the upper Chesapeake Bay.

When considering the costs of control and the additional economic losses due to decreased utilization of water resources, *Hydrilla* is a multimillion dollar problem. Because it is rooted to the bottom, *Hydrilla* is able to obtain its nutrition from bottom sediments as well as from the overlying water (Barko and Smart 1980, Barko 1982, Steward 1984; see also the section herein

on nutrition). The dioecious biotype of *Hydrilla* is able to dominate an entire body of water rapidly through its very efficient methods of vegetative reproduction. Fragments that break loose from established colonies sink to the bottom, become rooted, and form new colonies. Additionally, dioecious *Hydrilla* produces tuberlike propagules deep within sediments, which enable it to survive adverse environmental conditions including low temperatures, desiccation, and even herbicide applications (Mitra 1964, Steward 1969).

Dioecious *Hydrilla* has unique physiological characteristics that provide it with a competitive advantage over native species. It is able to utilize both dissolved carbon dioxide ($CO_2$) and bicarbonate ($HCO_3^-$) as carbon sources in photosynthesis (Van et al. 1976). It has a low light requirement, enabling it to grow in great depths of water (Bowes et al. 1977). Furthermore, *Hydrilla* is able to start utilizing carbon early in the day, thus potentially reducing the supply of carbon available to other species (Van et al. 1976).

The foremost characteristic enabling *Hydrilla* to be so successful in colonizing new areas is its ability to regrow from tubers. Tuber formation in the dioecious biotype has been observed to occur within 19 days after the planting of apical fragments (Steward, unpublished*). Tubers are produced in this biotype in response to short days (Van et al. 1978); recent studies indicate this is also true for the monoecious biotype (Steward and Van 1984).

## Influence of Vegetation on Aquatic Habitat

Aquatic macrophytes comprise an integral part of freshwater and estuarine systems and influence physical, chemical, and biological conditions. They are a dynamic component of the environment, with biomass and areal cover changing seasonally and in response to climatological events. The physical presence of stems, leaves, and roots influences currents, water depths, and deposition and erosion of sediments. Aquatic macrophytes create structural complexity within habitats by providing refuge and a substratum for a variety of organisms. Aquatic macrophytes can be direct and indirect sources of food for fish, ducks, and wading birds. Through normal processes of growth, senescence, and decomposition, aquatic macrophytes influence dissolved oxygen

---

* Dr. Kerry Steward, US Department of Agriculture, Ft. Lauderdale, Fla., 1984.

levels, bicarbonate equilibria, and quantities of particulate and dissolved organic matter.

Biological considerations

In aquatic systems, vegetation can be a source of organic matter directly or as detritus (dead vegetation) approximately equal to that from terrestrial plants (Westlake 1975 and references cited therein). Particulate organic matter forms a food base that is available throughout the year for a variety of invertebrate organisms, including filter feeders (Malmquist et al. 1978, Minshall 1967, Wallace et al. 1977) and deposit collectors (Cummins 1973, Cummins and Klug 1979, Lamberti and Moore 1984). These organisms in turn provide food for fish and other large animals.

Aquatic macrophyte communities composed of plants with different growth rates, stem and leaf configurations, etc., are structurally diverse and provide valuable habitat for many aquatic organisms throughout the year. In general, invertebrate diversity correlates positively with the density of aquatic vegetation (Gerking 1962, Nichols 1974). Killgore (1979) determined that in a stand of *Hydrilla*, the greatest invertebrate diversity occurred in the upper one meter of water where vegetation was thickest. Similarly, Morin and Kimball (1983) reported that periphyton were more abundant on the upper stems of watermilfoil (*Myriophyllum heterophyllum*) than on the lower portions of the plant.

A major contribution of plant communities to aquatic systems is the provision of colonization sites for other organisms such as snails, aquatic insects, protozoans, periphyton, bacteria, and fungi. In addition, amphipods, cladocerans, copepods, and other microcrustaceans are commonly collected on aquatic plants (Pennak 1953). Without the presence of structure (i.e., macrophytes), many invertebrates could not search for prey without escaping predation.

Pennak (1971) reported that a stream with thick growth of rooted aquatic plants had a standing crop of invertebrate biomass 3 to 10 times greater than a similar stream lacking aquatic vegetation. Minshall (1984) suggested that the major factor responsible for the high densities of attached invertebrates in vegetated areas is the increased surface area afforded by plants. Surface area varies among macrophyte species. Keast (1984) reported that invertebrate density (mainly Chironomidae) was least on wild celery (*Vallisneria americana*) and greatest on pondweed (*Potomogeton robbinsii*) and Eurasian watermilfoil

(*Myriophyllum spicatum*). Ball and Hayne (1952) reported that differences in invertebrate densities can result also from variations in water depth and macrophyte density.

The presence of submersed aquatic vegetation can produce a dramatic increase in invertebrate standing crop over that found in nonvegetated areas. In an oligohaline area of the Lower Hudson River, Menzie (1980) found that, from May to August, the invertebrates living on plants comprised 16 to 35 percent of the total number of invertebrates living in the sampling area (the remainder of the invertebrates lived in the bottom sediments). During the same time period, he also assessed larval chironomid (one of the most important invertebrate groups in tidal areas) biomass and found that the chironomid biomass on the plants made up 50 percent of the total chironomid biomass in the sampling area. Balciunas (1982) found a wide array of invertebrate taxa, including 64 insect taxa, on Eurasian watermilfoil within the United States. He found gastropods and chironomids to be the most common macroinvertebrates on these plants. Similarly, Martin and Shireman (1976) sampled the invertebrate epifauna present on *Hydrilla* in Florida and found chironomids and gastropods to be the most common macroinvertebrates present.

In addition to providing substrate for invertebrates, the presence of submersed aquatic vegetation results in enhanced invertebrate densities in the bottom substrates of plant beds as well. Menzie (1980) found the biomass of chironomids in the sediments of a plant-filled cove to be about eight times that of neighboring nonvegetated areas. Watkins et al. (1983) reported that the infaunal invertebrate numbers in the sediment of a *Hydrilla* bed were approximately four times the numbers present in the sediments of nearby nonvegetated areas. In rivers, a macrophyte bed acts like a filter, removing fine and coarse particulate matter that subsequently becomes incorporated into sediments (Greg and Rose 1982 and references cited therein). The diversity of benthic invertebrates is usually greater at vegetated sites because of sediment stability and the presence of organic matter as a source of food (Brouha and von Geldern 1979). Egglishaw (1964, 1969) found a positive correlation between amount of detritus (i.e., dead vegetation) and number of bottom-dwelling organisms.

The best available data concerning the benthic macroinvertebrate fauna of the freshwater tidal portion of the Potomac were obtained in a sampling program carried out by the USGS. This sampling was conducted from the autumn

of 1977 through the summer of 1979. One of the Potomac River sampling stations was located at river mile 89 in the Hunting Creek area, near what is now the center of the *Hydrilla* infestation. The most common infaunal macroinvertebrates collected by the USGS at river mile 89 included the tubificid oligochaetes *Limodrilua hoffmeisteri, L. cervix, L. udekemianus, L. claparedianus, Branchiura sowerbyi*, and *Ilyodrilus templetoni*; the Asiatic clam *Corbicula fluminea*; a sphaeriid clam *Musculium transversum*; and chironomid larvae belonging to the genus *Chironomus*. *Corbicula* first appeared in the Potomac River in 1975 (Dresler 1980) and is especially common in the area of the *Hydrilla* infestation. The bulk of the invertebrate taxa present in this area consisted of oligochaetes (both tubificid and naidid species), sphaeriid clams, and chironomids. Four gastropod (snail) species were also collected, along with six species of unionid clams (*Anodonta cataracta, A. implicata, Elliptio complanata, Lampsilis ochracea, L. cariosa, and L. ventricosa cohongoronta*).

The tidal freshwater benthic infauna of the Potomac is similar to that of other large tidal rivers flowing into Chesapeake Bay. Diaz and Boesch (1977) reported that the benthic community of the freshwater tidal area of the James River was numerically dominated by *C. fluminea, Limnodrilus* spp., *I. templetoni*, and *Coelotanypus* (Diptera:Chironomidae). *Corbicula fluminea, Limnodrilus* spp., and *I. templetoni* were among the most abundant benthic animals in the Potomac, and *Coelotanypus*, while not quite as common as the other taxa, was nevertheless present.

The increased density and diversity of aquatic organisms associated with aquatic plant communities has beneficial effects on fish. Killgore (1979) reported that largemouth bass and other game species were usually concentrated in *Hydrilla* beds located in shallow water. Holland and Lester (1984) found that average catches of northern pike from areas with submersed vegetation were more than 10 times greater than from sites with no vegetation. Davis and Hughes (1971) noted that angling success was greatest in sites with brush and standing timber, and, as a result, resource managers endeavor to increase fish production by encouraging plant growth and placing brush, trees, etc., in bodies of water.

Mittelbach (1981) and Hall and Werner (1977) demonstrated that open-water habitat can be risky for small bluegills due to predation by larger fish. Small fish generally avoid areas if there is danger of predation by larger

fish, even where invertebrate food sources are abundant. Juvenile bluegills consume invertebrate prey within vegetative cover, but ignore the same in open water. Laughlin and Werner (1980) reported that numbers of smaller sized longear sunfish and bluegill were positively correlated with height of vegetation and that few adults of either species used areas devoid of aquatic plants.

While the physical presence of plants may enhance aquatic habitats, an overabundance of plants can have negative effects. If vegetation or other structure is too dense, predatory organisms cannot easily find and capture their prey. If fish are unable to capture food or else expend large amounts of energy while searching for prey, their growth rates and physical condition are adversely affected. Relatively high densities of simulated aquatic vegetation (674 shoots/$m^2$) negatively influenced the feeding efficiency of killifish (*Fundulus heteroclitys*) (Heck and Thoman 1981). In a similar study, Savino and Stein (1982) determined that a density of 250 shoots/$m^2$ was required to diminish the ability of largemouth bass to capture prey. Colle and Shireman (1980) reported that the condition of harvestable sized largemouth bass was affected when *Hydrilla* density exceeded 30-percent coverage. These workers found that juvenile largemouth bass were better able to capture small food items; condition factors of this size class were only affected when plant densities exceeded 50-percent coverage. Wiley et al. (1984) reported that optimal macrophyte standing crop was no more than 52 grams dry weight/$m^2$ in central Illinois ponds dominated by pondweed (*Potomogeton crispus*) and bushy pondweed (*Najas flexis*).

The presence of aquatic vegetation can also directly influence fish reproduction. Fish that broadcast their eggs over aquatic vegetation or tree roots include northern pike, carp, goldfish, and goldenshiner. While nest builders (sunfishes, largemouth bass, crappies, rock bass, warmouth, bowfin, and most bullheads) lay eggs on mud, sand, or silt, they usually choose sites with vegetation (Lagler et al. 1962). In addition to providing a substrate for eggs, submersed vegetation provides cover for immature fish (considered above).

The use of *Hydrilla* as a food source has also been observed. Grass carp (white amur) and *Tilapia* are exotic species of freshwater fish that feed exclusively on *Hydrilla* and other aquatic plants, although many native freshwater fish also ingest aquatic vegetation. Hardin (1982) found *Hydrilla* in

stomachs of bluegill, green sunfish, and small largemouth bass. While some of this vegetation was probably consumed inadvertently during forage for invertebrates, the plant consumption still contributed to overall nutrition. Channel catfish (*Ictularis punctatus*) less than 12 in. long have been reported to feed on *Hydrilla*; additional consumers of this and other plants include other species of catfish and largemouth bass.

Waterfowl such as mallards, teal, and black ducks feed extensively on succulent plants at the water surface or in shallow areas. In a study conducted by Anderson and Low (1976), it was determined that mallards, canvasbacks, and coots removed 11.4 to 75.8 percent of sago pondweed (*Potamogeton pectinatus*) foliage and 30.8 to 67.1 percent of its tubers in experimental plots. Equivalent information is not available specifically for *Hydrilla*; however, its tubers are likely to provide a palatable food source for waterfowl. Diving ducks and shorebirds such as herons and sandpipers feed on small fish, amphibians, and invertebrates that are associated with emergent or submersed plants. Plant-associated invertebrates are an important dietary constituent for many waterfowl, especially during the breeding season and periods of molting (Krapu and Swanson 1975, Serie and Swanson 1976).

Physical/chemical considerations

The presence of macrophytes has an important influence on hydraulic conditions and sedimentation in aquatic systems. A thick mat of vegetation may block circulation of warm water to lower depths. Dale and Gillespie (1977) demonstrated a correlation between macrophyte biomass and steepness of the temperature gradient from water surface to substrate. Hillebrand (cited in Edwards 1968) reported water level increases of 2 to 3 times ambient in shallow reaches of small rivers due to impeded flow by dense stands of aquatic plants. Submersed aquatic vegetation can reduce current velocities and turbulence, resulting in an increase in sedimentation and a decrease in erosion of bottom sediments (Sculthorpe 1967, Brown 1975, Greg and Rose 1982). In the study of Harlin and Thorne-Miller (1982), vegetated plots accreted 2.5 cm of sediments, while plots devoid of vegetation eroded 1.0 cm. The effects of increased sedimentation may include a reduction in suspended solids (turbidity) as well as changes in both riverine and tidally driven circulation patterns. A decrease in turbidity in the Potomac coupled with the stabilization of bottom sediments may favor the spread of *Hydrilla* or perhaps the reestablishment of native vegetation.

Aquatic vascular plants may increase dissolved oxygen levels during the day because of photosynthetic activity (Edwards 1968). The contribution of vascular plants to oxygenation is greater in medium-sized rivers that are characterized by less turbulence (resulting in areal oxygenation) than in smaller streams (Wetzel 1975). Periphyton (attached algae) on plant stems and leaves is another important source of dissolved oxygen in aquatic systems. In contrast, respiration in dense plant beds may occasionally exceed photosynthesis because of self-shading, resulting in the removal of dissolved oxygen from the water. Macrophyte respiration at night can also consume large quantities of dissolved oxygen.

Water samples taken from aquatic plant beds frequently display elevated pH, reduced alkalinity, and absence of free dissolved bicarbonate because of photosynthetic activity (Patten 1956, Swindale and Curtis 1957, Kimball and Kimball 1977). The increase in pH and decrease in free $CO_2$ due to $HCO_3^-$ uptake may unfavorably impact vegetation, depending on the availability of free $CO_2$. These changes may result in the further spread of species able to utilize $HCO_3^-$ or species capable of forming dense canopies at the water surface (Adams et al. 1974), where exchange of atmospheric $CO_2$ can result in the attainment of nuisance levels of plant growth.

Sites with vegetation exhibit high levels of dissolved and particulate organic matter resulting from senescence and degradation of stems and leaves. In addition, aquatic plants play a significant role in the cycling of minerals. Calcium, phosphorus, magnesium, iron, and other minerals present in the earth's crust enter rivers by way of overland flow and groundwater seepage. Plants incorporate these elements, which can then be passed throughout the food web from herbivores to carnivores before reentering the soil or water.

## Environmental Factors Affecting Growth

### Salinity tolerance

The successful colonization of the upper Potomac by the monoecious biotype of *Hydrilla* (Steward et al. 1984) poses a threat to the resources of Chesapeake Bay, since floating stem fragments carried downstream are capable of entering the bay. The ability of *Hydrilla* to become established in the bay will depend on the plants' tolerance to salinities of the bay waters. Paschal

et al. (1982) reported salinities during 1978-1981 at the mouth of the river (in the vicinity of Lookout Point) to range from 7 to 15 ppt in the spring and 13.5 to 23 ppt in the fall.

Most studies dealing with salinity tolerance of submersed aquatic plants have been conducted with marine species and usually have been concerned with effects of salinity on species distribution (Brock 1982a, 1982b; Brock and Lane 1983; Howard-Williams and Liptrot 1980; Mayer and Low 1970; Phillips et al. 1983; Verhoeven 1975). Very few investigations have been conducted with freshwater plants, since this group appears to be generally intolerant to high salinity.

The environmental effects of salt from highway runoff on *Potamogeton alpinus* have been investigated by Rabe et al. (1982). Other studies have investigated the effects on submersed freshwater vegetation of salinity from alkaline soils (Kollman and Wali 1976). McGahee and Davis (1971), using seawater or mixtures with artificial seawater, found photosynthesis and respiration of *Myriophyllum spicatum* to be unaffected at 16 ppt salinity. Kadono (1982) observed *Hydrilla* growing in waters of 6.5 ppt salinity in Japan. Howard-Williams and Liptrot (1980) studied the distribution of submersed species in a brackish lake system in South Africa and found that *Potamogeton pectinatus* and *Chara* did not grow where salinities exceeded 20 ppt; however, these plants survived several months exposure to 16 ppt. Verhoeven and Vierssen (1978) reported an upper tolerance of 15 ppt for *Potamogeton*. Davis et al. (1974) observed a relationship between salt toxicity and calcium concentrations in treatment solutions. Forney and Davis (1971) observed no significant effect on growth of *Vallisneria americana* up to 6 ppt salinity from artificial seawater.

There appear to be only two studies dealing specifically with salinity tolerance of *Hydrilla*. In a laboratory study, Haller et al. (1974) bioassayed dioecious *Hydrilla* against several treatments of diluted seawater. They observed no growth beyond 6.7 ppt salinity, while the growth of *Najas guadulepensis* and *Vallisneria americana* was inhibited beyond 10 ppt and *Myriophyllum spicatum* beyond 13.3 ppt. Steward and Van (1984), in preliminary laboratory studies of monoecious and dioecious biotypes, reported a threshold level of 13 ppt for both biotypes in trials with diluted seawater. After 6 weeks, biomass in 14 ppt seawater was reduced 29 percent of controls. There is a clear discrepancy in results reported in these two studies, necessitating

a more rigorous examination of salinity tolerance in *Hydrilla*. At present, based on field observations, it is assumed that *Hydrilla* will not survive in the Potomac at salinities greater than 5 ppt.

Nutrition

It was once widely held that nutrients were absorbed almost exclusively from the surrounding water by the shoots of submersed aquatic macrophytes, and that roots in sediments functioned only as anchoring devices (refer to historical review in Sculthorpe 1967). The role of water versus sediment in the nutrition of these plants remains a subject of continuing debate (cf. Waisel et al. 1982). However, it is now generally agreed that under many circumstances two very important elements, nitrogen and phosphorus, are mobilized primarily from sediments via root uptake (refer to literature reviews in Barko and Smart 1981a, Denny 1980, Huebert and Gorham 1983, Smart and Barko 1985a). The role of sediment as a direct source of nitrogen and phosphorus is ecologically quite significant, since these two elements, due in part to their rapid removal from solution by microorganisms, are normally very low in concentration in available forms in the open water of many aquatic systems. The availability of micronutrients to submersed macrophytes in the water is usually quite low, due not only to removal from solution by microorganisms but also to their precipitation with oxyhydroxide complexes (Wetzel 1983). Nitrogen, phosphorus, and micronutrients are relatively abundant in available forms within most sediments, from which they can be mobilized effectively by submersed macrophytes (Huebert and Gorham 1983, Smart and Barko 1985a). Other biologically important elements (calcium, magnesium, potassium, sodium, and sulfur) can be obtained from either sediment or overlying water (cf. Denny 1980, Smart and Barko 1985b); considering the normally high concentrations and conservative nature of these elements in the open water of most aquatic systems, it is unlikely that their availability ever directly limits the growth of submersed macrophytes. Clearly, the dual source (i.e., sediment and water) of nutrients to rooted submersed macrophytes provides a nutritional advantage over nonrooted vegetation.

The literature related to submersed macrophyte nutrition certainly stresses the importance of sediments, but it is biased somewhat toward lacustrine (nonflowing water) environments. In riverine systems, greater hydraulic exchange, generally coarser sediments, and perhaps lesser competition with phytoplankton for nutrients may favor proportionately greater nutrient uptake

from water than from sediments. Low Secchi disk transparencies and frequently high aqueous chlorophyll concentrations in the Potomac (Paschal et al. 1982) indicate a nutrient rich environment. These observations, in combination with the paucity of evidence in the literature indicating specific nutrient limitation in submersed macrophytes, suggest that nutrients in the Potomac may generally exceed the physiological requirements for growth of all submersed macrophytes including *Hydrilla*.

Sediment tolerances and requirements

Much of the recent research on the nutrition of submersed macrophytes was stimulated by earlier experimental accounts and observations of sediment-related variations in macrophyte growth and distribution (Pond 1905, Pearsall 1920, Misra 1938, Moyle 1945). Whereas variations in macrophyte growth on different sediments may in some cases involve nutrition, this has not been unequivocally demonstrated (cf. Barko and Smart 1983). Alternatively, it has been suggested that the principal influence of sediments on submersed macrophytes is due to physical texture rather than chemical composition (Sculthorpe 1967). Texture is important in relation to the rooting depth of species with different abilities to penetrate sediment (Denny 1980), and it may influence rooting success in particular conditions of water flow (Haslam 1978).

Among the numerous properties of sediments potentially affecting macrophyte growth and distribution, organic and inorganic constituents formed anaerobically have received the greatest attention. With the addition of organic matter, an aquatic environment may experience a high demand for dissolved oxygen and the development of chemically reducing (i.e., anaerobic) conditions. Sulfide, an anaerobic product found in sediments, originates from microbial degradation of organic sulfur and the reduction of oxidized inorganic sulfur compounds. Soluble sulfides, including $S^{=}$, $HS^{-}$, and $H_2S$, are considered highly toxic to plants and other soil organisms (Sanderson and Armstrong 1980). Although large quantities of sulfide are frequently produced in anaerobic aquatic environments, the concentration of water-soluble hydrogen sulfide may in fact be quite small (Ponnamperuma 1972), since it is readily oxidized by ferric hydroxide or geothite or by the growing roots of aquatic plants. High concentrations of soluble $H_2S$ normally require the sustained input of sulfate in combination with the reducing potential of organic matter. Such conditions are likely to occur in saline reaches of the tidal Potomac.

Organic products of anaerobic metabolism include volatile fatty acids, methane, ethylene, phenols, and alcohols (Yoshida 1975, Drew and Lynch 1980). Organic compounds in general have been demonstrated in the laboratory to inhibit the growth of dioecious *Hydrilla* among other submersed aquatic vegetation (Dooris and Martin 1981, Barko and Smart 1983). By comparing the distribution of macrophytes in polluted and unpolluted waters, Kullberg (1974) and Ozimek (1978) suggested that an increase in organic loading results in a loss of macrophytes.

Metals have been shown to affect the natural distribution of many wetland plant species (Martin 1968; Jones 1971, 1975). Chemical reduction of iron and manganese commonly occurs under anoxic sediment conditions. High concentrations of reduced iron and manganese may be contained in the interstitial waters of Potomac River sediments, since large quantities of these metals have been detected in the bulk phase of Potomac River sediments (Paschal et al. 1982). Although high concentrations of iron and manganese are normally considered toxic to plants (Sanderson and Armstrong 1980), some submersed macrophyte species grow best in reduced sediments containing significant amounts of soluble manganese (Pulich 1982). Soluble iron can inhibit the growth of submersed aquatic vegetation by interfering with sulfur metabolism or by limiting the availability of phosphorus (Jones 1975).

It is possible that sediment composition will play an important role in influencing the distribution of monoecious *Hydrilla*. However, no detailed information on composition of sediment in the shallows of the Potomac, particularly with respect to the presence or absence of organic constituents, is available. Furthermore, the sensitivity of monoecious *Hydrilla* to sediment properties is unknown.

Light and temperature

The availability of light for photosynthesis in submersed macrophytes is an important factor in controlling growth rates and areal distribution. Based on an extensive survey of Scottish freshwater lochs, Spence (1967) proposed that the zonation of a variety of submersed macrophyte species along a gradient in water depth was determined primarily by light regime. In related investigations, Spence and Chrystal (1970a, b) demonstrated a greater photosynthetic capacity in deep water compared to shallow water for *Potamogeton* species and suggested that the natural depth distribution of these species was linked to shade tolerance. Light plays an important role also in seasonal

changes in macrophyte dominance and in interspecific competition (refer to later section on competition in this chapter). Westlake (1981) provided evidence that reductions in irradiance may eliminate some species and allow other species to replace them over months or years.

In many aquatic systems, but particularly in riverine systems and estuaries, the depth distribution of submersed macrophytes may be severely limited by inadequate penetration of light associated with the presence of high concentrations of suspended sediment. Reductions in water clarity also occur during phytoplankton blooms, which can appreciably reduce the growth of submersed macrophytes (Jupp and Spence 1977). Suppressed growth of submersed macrophytes due to shading by epiphytes (attached algae) has also been reported (Phillips et al. 1978, Sand-Jensen and Søndergaard 1981). Epiphytes apparently reduce leaf photosynthesis by acting as a barrier to carbon uptake as well as by shading (Sand-Jensen 1977). In some systems such as the lower Chesapeake Bay, regulation of epiphyte abundance by invertebrate grazers may play a major role in determining the growth potential and distribution of submersed macrophytes by improving the availability of light at leaf surfaces (Orth and Montfrans 1984).

Morphological flexibility is important in determining species success in low light environments. For example, submersed macrophytes that are capable of elongating to the water surface and/or forming a surface canopy, such as *Myriophyllum spicatum* and *Hydrilla verticillata* (dioecious variety), may grow to greater depths and have a competitive advantage over species possessing a low-profile growth form, such as *Vallisneria americana* (Haller and Sutton 1975, Titus and Adams 1979, Barko and Smart 1981b). Past periods of dominance by *Myriophyllum spicatum* in the upper Chesapeake Bay (Bayley et al. 1978) may partially reflect the exceptional ability of this species to form a surface canopy under conditions of limited water transparency (Adams and McCracken 1974, Adams et al. 1974). An equivalent or greater ability of *Hydrilla* to form a surface canopy likely accounts for its complete dominance in many aquatic systems of Florida. However, based on personal observations and limited laboratory data (Barko unpublished*), monoecious *Hydrilla* does not appear to possess the shoot-elongation properties and canopy-forming ability

---

* Dr. John W. Barko, US Army Engineer Waterways Experiment Station, Vicksburg, Miss., 1985.

of the dioecious biotype. Nevertheless, *Hydrilla* appears to be growing very densely in some parts of the Potomac. Its ability to compete with other species in these areas is presently unknown and needs to be defined.

Changes in the species composition of submersed macrophyte communities due to thermal alterations (Anderson 1969, Allen and Gorham 1973) suggest that temperature may be as important as light in influencing competitive relations among coexisting species. In Florida, *Hydrilla* (dioecious) is apparently able to displace *Egeria densa* seasonally, because of the greater thermal tolerance of the former compared to the latter (Barko and Smart 1981b). The biogeography of many submersed macrophytes suggests a temperature effect on latitudinal distribution (Sculthorpe 1967). Until recently, based on studies involving the dioecious strain (Barko and Smart 1981b and literature cited therein), *Hydrilla* was considered to be primarily a subtropical species. The Potomac River is clearly not a subtropical environment, but again it is critical to emphasize that the information available from studies involving the dioecious biotype of *Hydrilla* is not strictly applicable to the monoecious biotype. The thermal tolerance of the latter is not presently known; without this information, no reliable predictions of the northward advancement of monoecious *Hydrilla* can be made.

Water chemistry

Considerable information has been collected in attempts to relate the distribution of submersed aquatic macrophytes to various water chemistry parameters. In addition to salinity (considered independently in Chapter I), the most often considered parameters include alkalinity, calcium (hardness), pH, nutrient status or degree of eutrophication, conductivity, and total dissolved solids (Moyle 1945, Hutchinson 1975). These parameters are often closely interrelated (Hutchinson 1957), thus complicating the determination of specific mechanisms involved in determining species distribution (Hutchinson 1970, 1975). Sites differing in water chemistry are likely to differ in other environmental factors (notably sediment composition) as well (Pearsall 1920, Misra 1938, Moyle 1945).

Effects of nutrient enrichment on submersed macrophytes are generally caused by an increase in phytoplankton and epiphyte biomass (Mulligan and Baranowski 1969, Ryan et al. 1972, Mulligan et al. 1976, Jupp and Spence 1977, Phillips et al. 1978, Sand-Jensen and Søndergaard 1981, Twilley et al. 1985; also see discussion of light and competition in the preceding section of this

chapter). Alkalinity, $Ca^{++}$, and pH can affect the distribution of aquatic plants primarily through their influence on inorganic carbon acquisition. Although there is a considerable body of information available on the distribution of submersed macrophytes in relation to alkalinity, Ca, and pH (Moyle 1945; Spence 1967; Hutchinson 1970, 1975; Seddon 1972; Hellquist 1980; Kadono 1982), associated mechanisms are poorly understood (Smart and Barko 1985b). Submersed macrophytes classified as "soft-water species" depend primarily on free $CO_2$, while those classified as "hard-water species" can utilize $HCO_3^-$ as well (Kadono 1980).

A major influence of water chemistry on submersed macrophyte species is related to the ability of these plants to utilize $HCO_3^-$ in photosynthesis (Hutchinson 1975). The dioecious *Hydrilla* common in Florida has the ability to utilize $HCO_3^-$ (Van et al. 1976), and preliminary evidence obtained in the Environmental Laboratory at WES and by the US Department of Agriculture laboratory in Fort Lauderdale suggests that the monoecious biotype can likewise utilize $HCO_3^-$.

## Reproduction

In *Hydrilla*, vegetative (asexual) reproduction is the primary mechanism for propagation as well as perennation and dispersal (Pieterse 1981). Modes of asexual reproduction in *Hydrilla* may be separated into two major types: "indeterminant," such as fragments of stems and rhizomes, which can produce new plants within a few days during the normal growing season, and "determinant," including axillary turions, subterranean turions (tubers), and root crowns. (Note: turions are dormant buds.) The determinant types of reproduction represent physiological commitments to specialized organs and are usually the ones on which the plant relies to overwinter or withstand periods of desiccation.

Any fragment of a stem that contains at least one node and intact leaves is capable of starting a new plant. However, experimental results have shown that fragments having two or more nodes have a higher frequency of new shoot production (Anderson 1984). In dioecious *Hydrilla*, new shoots are formed on two-node fragments at temperatures above 15°C when exposed to light. The minimum light requirements and temperature for new shoot production on fragments of monoecious *Hydrilla* are not known. Growth rates of the new shoots are

temperature and light dependent and can reach 0.5 to 1 cm per day. Formation of roots usually follows new shoot formation by several days. Thus, small pieces of *Hydrilla* having two or three nodes can effectively propagate and disperse the plant, especially in flowing water where the floating fragments can be dispersed several miles a day (Anderson and Dechoretz 1982). Even under conditions not conducive for new shoot production, two-node sections of *Hydrilla* can remain viable for weeks and later initiate new shoots under elevated light and temperature.

In dioecious *Hydrilla*, rhizomes (and root crowns) seem to be able to overwinter and thus serve as perennating organs. Field samples from monoecious populations in the Potomac River suggest that this biotype does not have perennating rhizomes. Preliminary observations on monoecious *Hydrilla* grown outdoors in Florida support this conclusion (Steward, personal observations*). It should be noted, however, that perennating rhizomes are likely to be of minor importance compared to other modes of reproduction (i.e., tuber formation, fragmentation, etc.) in this species.

Newly emerged rhizomes produce a locus or nodal area from which roots are initiated, resulting in a distinct root crown from which several more shoots and roots grow (Yeo et al. 1984). These arrays of root crowns serve as establishment points for new tuber formation.

Two types of turions are produced in *Hydrilla*: axillary turions (produced on aboveground parts) and subterranean turions (commonly called "tubers"). Microscopical examination shows that the two types of turions are anatomically and developmentally similar (Yeo et al. 1984); however, tubers are generally firmer structures and are more resistant to mechanical disruption. In contrast, the axillary turions are released in the water column and thus can be dispersed in flowing water.

Field and laboratory studies have shown that dioecious plants are induced to form tubers under short-day conditions (Haller 1976, Haller et al. 1976, Van 1978, Van et al. 1978, Bowes et al. 1979). Less is known about what influences production of axillary turions, but some evidence points to differences in biotype (Pieterse 1981) and shifts from warm to cold temperature (e.g., from 25° to 15°C) (Sastroutomo 1980). Exposure to low concentration

---

* Dr. Kerry Steward, US Department of Agriculture, Ft. Lauderdale, Fla., 1984.

of the plant growth regulator abscisic acid (ABA) also can induce turion formation (Van et al. 1978, Klaine and Ward 1984). Studies on source/sink relationships in other plants suggest that temperature and photoperiod may interact in turion formation (Melis 1984).

Short days also induce tuber formation in the monoecious plants, but within 2 to 3 weeks compared to 8 to 12 weeks in the dioecious plants (Steward and Van 1984, Spencer and Anderson 1985). The importance of this is evidenced in the fact that tuber formation has been observed in the Potomac River by the end of June (Anderson, unpublished observations*).

Tuber induction in the dioecious plants is photoreversible by red or white light (Van et al. 1978, Klaine and Ward 1984). This is typical of photochrome-mediated systems (Song 1984) and is undoubtedly true for the monoecious plant as well. The applicability of these findings to the monoecious biotype and the possible utility of photoreversal as a control method have not been determined.

Dioecious *Hydrilla* tubers require temperatures above ca. 12°C (either in light or dark) for sprouting; optimal range is between 22° and 28°C (Miller et al. 1976). The optimal and minimum temperatures for germination of monoecious tubers has not yet confirmed, but they have sprouted at 15°C in the laboratory (Steward and Van 1984) and in the fall at Davis, Calif., in water temperatures around 15° to 17°C (Anderson unpublished observations*).

Laboratory studies have shown that gibberellic acid can stimulate germination (and subsequent elongation), while abscisic acid (ABA) inhibits germination (Steward 1969, Anderson 1984). Tubers can be killed by contact with the soil fumigant metham (Vapam = sodium methyldithiocarbamate) when it is applied as a soil drench or via subsurface injection (Steward 1969).

Postgermination vigor is probably related to tuber size, particularly for extremely small and large tubers. Given the smaller size of monoecious tubers (relative to dioecious ones), they may not be able to compete as well under suboptimal conditions. Likewise, if the monoecious tubers do not last more than one season, then this plant may behave essentially as an annual. If this is true, then management by prevention of tuberization could be a successful approach.

---

* Dr. Lars W. J. Anderson, University of California, Davis, Calif., 1984.

In general, plants in the family Hydrocharitaceae, of which *Hydrilla* is a member, do not produce large numbers of viable seed (Sculthorpe 1967). Consequently, seed production constitutes a minor form of reproduction. However, field samples and preliminary greenhouse studies have shown that the monoecious biotype can produce viable seed (Conant et al. 1984). Flower production in both the female dioecious and in the monoecious plants in the US is induced by short days (Yeo et al. 1984). Although considerable pollen is produced by the male flower, the chances of contact with the female flower and subsequent production are small (Cook and Luond 1982). Limited experimental crossings indicate that there is some potential for exchange of genetic material between the dioecious female plants, which are widely distributed in the US, and the newly established populations of the monoecious plant (Schwarzenbach 1945). Considering, however, the probable infrequency and stochasic nature of genetic exchange in *Hydrilla*, this process is unlikely to have an effect on its specific adaptability to the Potomac environment.

## Competition

### Influence of irradiance conditions and depth

Submersed plant species exhibit varying degrees of stem elongation (Goldsborough 1983). One of the adaptations of dioecious *Hydrilla* is its significant elongation potential, which positions photosynthetic tissues near the water surface. The ability to form a dense canopy confers a significant competitive advantage over phytoplankton, benthic algae, and slower growing higher plants such as *Vallisneria*, particularly in turbid systems (Haller and Sutton 1975). Based upon preliminary data and observations, the monoecious biotype of *Hydrilla* appears to be relatively less proficient at elongation and subsequent canopy formation.

Another aspect of competition for light is the ability of *Hydrilla* to photosynthesize at lower light levels than competitors. It now appears that the *Hydrilla* populations (dioecious) studied by Van et al. (1976) and the monoecious biotype (Van, personal communication*) have much higher $CO_2$ fixation rates at low light levels than any other species studied. Although the maximum rate of photosynthesis in *Hydrilla* at saturating irradiance is not as

---

* Dr. Thai K. Van, US Department of Agriculture, Ft. Lauderdale, Fla., 1984.

high as some species such as *Ceratophyllum*, it may be extremely successful in the low light field of turbid estuaries such as the Potomac River. Although *Hydrilla* has one of the highest maximum rates of dry matter production (11.5 $g/m^2/day$) among submersed species (Singh and Sahai 1977), it is uncertain as to whether these high rates can be maintained under estuarine conditions.

Influence of nutrient loading

The Potomac River estuary is one of the more eutrophic portions of Chesapeake Bay. Therefore, it is unlikely that competition for nutrients would influence the success of *Hydrilla* in this area (see earlier section on nutrition). However, nutrients may be important indirectly by promoting excessive algal growth on the leaves of macrophytes. Kemp et al. (1983) have suggested that epiphytic algal overgrowth is the major reason for the decline of the native submersed aquatics in Chesapeake Bay. Under these conditions, a species having the ability to reduce algal growth would have a competitive advantage over other species. Kulshreshtha and Gopal (1983) have reported the effects of allelopathic substances associated with *Hydrilla* populations in Jaipur, India. These phytotoxic substances apparently prevent higher plant species such as *Ceratophyllum* from growing in mixed stands with *Hydrilla* and may also benefit *Hydrilla* by reducing algal colonization on leaves resulting from high nutrient levels (refer to previous section on environmental factors).

Influence of salinity

Preliminary data related to the salinity tolerance of monoecious *Hydrilla* indicate that it could extend its present distribution into more saline areas of the Potomac River estuary (see earlier section on salinity tolerance). A possible deterrent to this expansion is likely to be competition with other submersed macrophyte species such as *Ruppia maritima*, *Potamogeton pectinatus*, and *Myriophyllum spicatum*, which are better adapted to saline conditions. In this regard, the exotic *Myriophyllum spicatum*, now in the Potomac and with a past history of dominance in Chesapeake Bay, may be a major competitor with *Hydrilla*. The ability of *Hydrilla* to compete with other species of the Potomac is unknown due to the present lack of field and laboratory data.

## Conclusions and Recommendations

### Habitat

It is apparent that the presence of submersed aquatic vegetation can dramatically increase invertebrate standing crop, both by furnishing a substrate for colonization by epifauna and by enhancing invertebrate densities within bottom sediments. In the Potomac River, where submersed aquatic vegetation can potentially occupy fairly large areas, macrophyte beds may contain a significant portion of the river's invertebrate numbers and biomass. Important Potomac River fishes such as the chain pickerel, white catfish, striped bass, and white perch all feed on invertebrates at some stage of their lives (Menzie 1980).

Strategies for management of *Hydrilla* in the Potomac River should be designed to account for beneficial as well as negative effects of *Hydrilla* on the system. Information requirements to determine these effects include the following.

a. Role of vegetation in the Potomac River determined by an analysis of species composition, percent cover, standing crop, and annual production.

b. Contributions of aquatic macrophytes in the Potomac: their importance as a sink for nutrients, a source for particulate matter, and a factor potentially affecting sedimentation rates.

c. Association of numbers and biomass of invertebrates with selected plant species and the underlying sediments in vegetated and nonvegetated sites, because these organisms contribute to fish production.

d. Analysis of fish densities, fish condition factors, and stomach contents of sport and commercial fish.

e. Relation of changes in macrophyte density and species composition to numbers of waterfowl and shorebirds.

The information described in the preceding subparagraphs would support a proper evaluation of the ecological importance of existing vegetation and the related significance of future changes in plant densities and coverage.

### Environmental factors

In order to assess the potential development of *Hydrilla* and other vegetation in the Potomac, it is necessary to determine the importance of environmental factors affecting macrophyte growth. Whereas a good deal of information is available on the ecology of submersed macrophytes, most of this

is based on lacustrine studies. Vegetation in riverine systems, particularly where influenced by tides as in the Potomac, has received far less attention. Assessment of short- or long-term responses of monoecious *Hydrilla* to conditions in the Potomac is further complicated by an inadequate understanding of its environmental tolerances and requirements. It is important to emphasize that information available from studies involving dioecious *Hydrilla* is not strictly applicable to the monoecious biotype.

The ability of monoecious *Hydrilla* to become established in the lower reaches of the Potomac River and in the Chesapeake Bay will depend, in part, on its tolerance to salinity. Preliminary evidence indicates an upper salinity-tolerance range of 6 to 13 ppt for *Hydrilla*,* which is within the range of salinities observed at the mouth of the Potomac River.

The Potomac River is a nutritionally rich environment, and the spread of monoecious *Hydrilla* is unlikely to be limited by low levels of nutrients. Conversely, excess nutrients may reduce the growth and distribution of *Hydrilla* by favoring highly competitive phytoplankton and epiphytic algae.

The distribution of monoecious *Hydrilla* is related to the influence of physical and chemical properties of underlying sediments. The growth and potential spread of monoecious *Hydrilla* will likely depend on its ability to utilize sediments of widely varying composition.

Monoecious *Hydrilla* seems to be more tolerant of low temperatures than the dioecious biotype. However, the specific temperature tolerance of monoecious *Hydrilla* has not been determined.

Light is likely to be the most important environmental factor limiting the growth and distribution of monoecious *Hydrilla* in the Potomac. Competition among submersed macrophyte species will be affected by differential abilities to cope with low light conditions.

The following information should be included in an assessment of the potential development of *Hydrilla* and other vegetation in the Potomac.

a. Specific salinity tolerance of monoecious *Hydrilla* in relation to the salinity tolerance of potential competitors including native vegetation.

b. Influence of nutrients (both in water and in sediment) on the growth of monoecious *Hydrilla*.

---

* Dr. Kerry Steward, US Department of Agriculture, Ft. Lauderdale, Fla., unpublished data, 1984.

c. Effects of texture, reduced substances, nutrients, salinity, and organic compounds on the potential distribution of monoecious *Hydrilla*.

d. Specific temperature requirements for growth and reproduction of monoecious *Hydrilla* and comparison with those of potential competitors. (A comparative assessment of the temperature requirements and tolerances of the two *Hydrilla* biotypes might allow a determination of their potential for coexistence.)

e. Capacity of monoecious *Hydrilla* to photosynthesize under low light conditions, because this biotype, as opposed to the dioecious biotype, seems to be limited in its ability to form a canopy.

f. Morphological as well as physiological adaptations of monoecious *Hydrilla* to low light conditions and comparison with those of potential competitors.

g. Effects of water chemistry on photosynthesis and growth on both *Hydrilla* biotypes, particularly the ability of monoecious *Hydrilla* to utilize bicarbonate in photosynthesis.

Reproduction

The rapid spread of *Hydrilla* is due primarily to its many efficient modes of vegetative reproduction (fragmentation and tuber and turion formation). The production of tubers and turions is important to *Hydrilla's* survival and distribution; control of *Hydrilla* might be achieved by inhibition of their formation, perhaps by extending the photoperiod or by using plant-growth regulators. The role of plant-growth regulators in *Hydrilla* tuber formation is not known.

*Hydrilla* fragments and turions are the structures that are the most likely means by which the plant will invade downstream areas. Salinity and interactions between sediment type, water temperature, and tuber formation will determine the limits of population expansion. It is not known if *Hydrilla* adjusts its tuber production with population density, but this information would be useful in predicting stand density and in assessing age of existing populations. The potential also exists for long-term adaptation of *Hydrilla*.

The following information should be included in an assessment of the potential spread of *Hydrilla* in the Potomac River.

a. Importance of vegetative reproduction in *Hydrilla* determined by the viability (longevity) of turions, tubers, and fragments.

b. Methods of inhibition of formation of tubers and turions. (Results could be useful in developing a practical nonpolluting method for management.)

c. Specific functions of plant-growth regulators in *Hydrilla*, possibly leading to selective management through interruption of normal growth and reproduction processes.

d. Potential for long-term adaptation of *Hydrilla* by sexual (seed) reproduction based on studies conducted on self-crosses with monoecious *Hydrilla* and crosses between monoecious and dioecious biotypes.

Competition

Dioecious *Hydrilla* has been found to be very successful in displacing native aquatic vegetation in many environments. This success is due to its ability to form a dense canopy that shades other species and to its prolific reproduction potential. In contrast, little information is available to evaluate the competitive nature of monoecious *Hydrilla*.

Evaluation of the competitive nature of dioecious *Hydrilla* requires the following information.

a. Competitiveness of dioecious *Hydrilla* over the gradient of salinities, nutrient loadings, and turbidities that characterize the Potomac River estuary.

b. Outcome of competition involving *Hydrilla* and native submersed macrophyte species as determined by small-scale bioassay experiments complemented with larger scale experiments under field conditions.

c. Effects of water chemistry on competition involving *Hydrilla* and native species under conditions representative of those occurring in different environments of the Potomac River.

## Contributors


| | |
|---|---|
| Dr. Lars W. J. Anderson | USDA/ARS Aquatic Weed Control Research<br>Botany Department<br>University of California<br>Davis, CA 95616 |
| Dr. John W. Barko* | US Army Engineer Waterways Experiment Station<br>ATTN: WESES-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. David C. Beckett | US Army Engineer Waterways Experiment Station<br>ATTN: WESER-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. Rex L. Chen | US Army Engineer Waterways Experiment Station<br>ATTN: WESES-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. Robert King | Department of Biology<br>Central Michigan University<br>Mt. Pleasant, MI 48859 |
| Dr. Andrew C. Miller | US Army Engineer Waterways Experiment Station<br>ATTN: WESER-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. R. Michael Smart | US Army Engineer Waterways Experiment Station<br>ATTN: WESES-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. J. Court Stevenson | Horn Point Lab<br>University of Maryland<br>Cambridge, MD 21613 |
| Dr. Kerry Steward | Aquatic Plant Management Lab<br>USDA, ARS<br>3205 SW College Avenue<br>Ft. Lauderdale, FL 33314 |


---

* Dr. Barko was primarily responsible for the development of this chapter and should be used as a point of contact regarding its content.

## Contributors


| | |
|---|---|
| Dr. Robert R. Twilley | Horn Point Lab<br>University of Maryland<br>Cambridge, MD 21613 |
| Dr. Thai K. Van | USDA/ARS<br>3205 SW College Avenue<br>Ft. Lauderdale, FL 33314 |

## References


Adams, M. S., J. E. Titus, and M. D. McCracken. 1974. Depth distribution of photosynthetic activity in a *Myriophyllum spicatum* community in Lake Wingra. Limnology and Oceanography 19:377-389.

Adams, M. S., and M. D. McCracken. 1974. Seasonal production of the *Myriophyllum* component of the littoral of Lake Wingra. Journal of Ecology 62:457-465.

Allen, E. D., and R. R. Gorham. 1973. Changes in the submerged macrophyte communities of Lake Wabamum as a result of thermal discharge. Pages 313-324. Proceedings of the symposium on the lakes of western Canada. University of Alberta, Edmonton, Canada.

Anderson, L. W. J., and N. Dechoretz. 1982. Growth, reproduction and control of *Hydrilla verticillata* (L.F.) Royle in an irrigated system in southwestern U.S. Proc. Europ. Weed. Res. Soc. 6th Symposium on Aquatic Weeds, pp. 54-61.

Anderson, L. W. J. 1984. Use of bioassays for allelochemicals in aquatic plants. Amer. Chem. Soc. Symp. Series 268, pp. 351-370.

Anderson, R. R. 1969. Temperature and rooted plants. Chesapeake Science 10:157-164.

Anderson, M. G., and J. B. Low. 1976. Use of sago pondweed by waterfowl on the delta marsh, Manitoba. The Journal of Wildlife Management 40:233-242.

Balciunas, J. K. 1982. Insects and other macroinvertebrates associated with Eurasian watermilfoil in the United States. Technical Report A-82-5, U.S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Miss.

Ball, R. C., and D. W. Hayne. 1952. Effects of the removal of fish population on the fish-food organisms of a lake. Ecology 33:41-48.

Barko, J. W. 1982. Influence of potassium source (sediment vs. open water) and sediment composition on the growth and nutrition of a submersed freshwater macrophyte [*Hydrilla verticillata* (L.f.) Royle]. Aquatic Botany 12:157-172.

Barko, J. W., and R. M. Smart. 1980. Mobilization of sediment phosphorus by submersed freshwater macrophytes. Freshwater Biology 10:229-238.

_____. 1981a. Sediment-based nutrition of submersed macrophytes. Aquatic Botany 10:339-352.

_____. 1981b. Comparative influences of light and temperature on the growth and metabolism of selected submersed freshwater macrophytes. Ecological Monographs 51:219-235.

_____. 1983. Effects of organic matter additions to sediment on the growth of aquatic plants. Journal of Ecology 71:161-175.

Bayley, S., V. D. Stotts, P. F. Springer, and J. Steenis. 1978. Changes in submerged aquatic macrophyte populations at the head of Chesapeake Bay, 1958-1975. Estuaries 1:73-84.

Blackburn, R. D., R. W. Weldon, R. R. Yeo, and T. M. Taylor. 1969. Identification and distribution of certain similar-appearing submersed aquatic weeds in Florida. Hyacinth Control Journal (USA) 8:17-21.

Bowes, G., T. K. Van, L. A. Garrard, and W. T. Haller. 1977. Adaptation to low light levels by *Hydrilla*. Journal of Aquatic Plant Management 15:32-35.

Bowes, G., A. S. Holaday, and W. T. Haller. 1979. Seasonal variation in the biomass, tuber density, and photosynthetic metabolism of hydrilla in three Florida lakes. Journal of Aquatic Plant Management 1:61-65.

Brock, M. A. 1982a. Biology of the salinity tolerant genus *Ruppia* L. in saline lakes in South Australia. I. Morphological variation within and between species and ecophysiology. Aquatic Botany 13:219-248.

_____. 1982b. Biology of the salinity tolerant genus *Ruppia* L. in saline lakes in South Australia. II. Population ecology and reproductive biology. Aquatic Botany 13:249-268.

Brock, M. A., and J. A. K. Lane. 1983. The aquatic macrophyte flora of saline wetlands in Western Australia in relation to salinity and permanence. Hydrobiologia 105:63-76.

Brouha, P., and C. E. von Geldern. 1979. Habitat manipulation for centrarchid production in western reservoirs. Pages 11-17 in D. L. Johnson, and R. A. Stein, editors. Response of fish to habitat structure in standing water. Special publication 6. North Central Division American Fisheries Society.

Brown, J. M. A. 1975. Ecology of macrophytes. Pages 244-262 in H. V. Jolly and J. M. A. Brown, editors. New Zealand lakes. Oxford/Auckland University Press, Wellington, New Zealand.

Colle, D. E., and J. V. Shireman. 1980. Coefficients of condition for largemouth bass, bluegill, and redear sunfish in hydrilla - infested lakes. Transactions of the American Fisheries Society 109:521-531.

Conant, R. D., T. K. Van, and K. K. Steward. 1984. Monoecious *Hydrilla* produces viable seeds in the United States. Aquatics 6:10.

Cook, C. D. K., and R. Luond. 1982. A revision of the genus *Hydrilla* (Hydrocharitaceae). Aquatic Botany 13:485-504.

Cummins, K. W. 1973. Trophic relations of aquatic insects. Annual Review of Entomology 18:183-206.

Cummins, K. W., and M. J. Klug. 1979. Feeding ecology of stream invertebrates. Annual Review of Ecology and Systematics 10:147-172.

Dale, H. M., and T. J. Gillespie. 1977. The influence of submersed aquatic plants on temperature gradients in shallow water bodies. Canadian Journal of Botany 55:2216-2225.

Davis, G. J., M. N. Jones, C. Z. Lunney, and G. M. Clark. 1974. Inhibition of sodium chloride toxicity in seedlings of *Myriophyllum spicatum* L. with calcium. Plant and Cell Physiology 15:577-581.

Davis, J. T., and J. S. Hughes. 1971. Effects of standing timber on fish populations and fisherman success in Bussey Lake, Louisiana. Pages 255-264 in G. E. Hall, editor. Reservoir fisheries and limnology. Special Publication No. 8. American Fisheries Society, Washington, DC, USA.

Denny, P. 1980. Solute movement in submerged angiosperms. Biological Revue 55:65-92.

Diaz, R. J., and D. F. Boesch. 1977. Habitat development field investigations, Windmill Point Marsh development site, James River, Virginia; Appendix C: environmental impacts of marsh development with dredged material: acute impacts on the macrobenthic community. Technical Report D-77-23, U.S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Miss.

Dooris, P. M., and D. F. Martin. 1981. Growth inhibition of *Hydrilla verticillata* by selected lake sediment extracts. Water Resource Bulletin 16:112-117.

Drew, M. C., and J. M. Lynch. 1980. Soil anaerobiosis, microorganisms, and root function. Annual Review of Phytopathology 18:37-66.

Dresler, P. V. 1980. The Asiatic clam, *Corbicula fluminea* (Muller), in the tidal Potomac River, Maryland. Estuaries 3:150-151.

Edwards, R. W. 1968. Plants as oxygenators in rivers. Water Research 2:243-248.

Egglishaw, H. J. 1964. The distributional relationship between the bottom fauna and plant detritus in streams. Journal of Animal Ecology 33:463-476.

_____. 1969. The distribution of benthic invertebrates on substrate in fast-flowing streams. Journal of Animal Ecology 38:19-33.

Elser, H. J. 1966. Status of aquatic weed problems in tidewater Maryland. Maryland Department of Chesapeake Bay Affairs, Annapolis, Maryland, USA.

Forney, D. R., and D. E. Davis. 1971. Effects of low concentrations of herbicides on submersed aquatic plants. Weed Science 29:677-685.

Gerking, S. D. 1962. Production and food utilization in a population of bluegill sunfish. Ecological Monographs 32:31-78.

Goldsborough, W. J. 1983. Response of the submerged vascular plant, *Potamogeton perfoliatus* L., to variable light regimes in an estuarine environment. M.S. Thesis. University of Maryland, College Park, Maryland, USA.

Gottschalk, J. S. Sept. 25, 1984. Taking care of the hydrilla "problem" might not be prudent. Article in the Arlington Journal, Arlington County Daily Newspaper.

Greg, W. W., and F. L. Rose. 1982. The effects of aquatic macrophytes on the stream microenvironment. Aquatic Botany 14:309-324.

Hall, D. J., and E. E. Werner. 1977. Seasonal distribution and abundance of fishes in the littoral zone of a Michigan Lake. Transactions of the American Fisheries Society 106:545-555.

Haller, W. T. 1976. *Hydrilla*: a new and rapidly spreading aquatic weed problem. Agricultural Experiment Station. 1-13.

Haller, W. T., and D. L. Sutton, and W. C. Barlowe. 1974. Effects of salinity on growth of several aquatic macrophytes. Ecology 55:891-894.

Haller, W. T., and D. L. Sutton. 1975. Community structure and competition between *Hydrilla* and *Vallisneria*. Hyacinth Control Journal 213:48-50.

Haller, W. T., J. L. Miller, and L. A. Garrard. 1976. Seasonal production and germination of hydrilla vegetative propagules. Journal of Aquatic Plant Management 14:26-28.

Hardin, S. 1982. Large-scale operations management test using the white amur at Lake Conway, Florida. Pages 217-228. Proceedings, 16th annual meeting, Aquatic Plant Control Research Planning and Operations Review. Miscellaneous paper A-82-3, US Army Engineer Waterways Experiment Station, Vicksburg, Mississippi, USA.

Harlin, M. H., and B. Thorne-Miller. 1982. Seagrass-sediment dynamics of a flood-tidal delta in Rhode Island (U.S.A.). Aquatic Botany 14:127-138.

Haslam, S. M. 1978. River plants. Cambridge University Press, Cambridge, England.

Heck, K. L. and T. Thoman. 1981. Experiments on predator-prey interactions in vegetated aquatic habitats. Journal of Experimental Marine Ecology 53:125-134.

Hellquist, C. B. 1980. Correlation of alkalinity and the distribution of *Potamogeton* in New England. Rhodora 82:331-344.

Holland, L. E., and M. L. Lester. 1984. Relationship of young-of-the-year northern pike to aquatic vegetation types in backwaters of the upper Mississippi River. North American Journal of Fisheries Management 4:514-522.

Howard-Williams, C., and M. R. M. Liptrot. 1980. Submerged macrophyte communities in brackish South African estuarine-lake system. Aquatic Botany 9:101-116.

Huebert, D. B., and P. R. Gorham. 1983. Biphasic mineral nutrition of the submersed aquatic macrophyte *Potamogeton pectinatus* L. Aquatic Botany 16:269-284.

Hutchinson, G. E. 1957. A treatise on limnology. I. Wiley. New York, New York, USA.

_____. 1970. The chemical ecology of three species of *Myriophyllum* (Angiospermae, Haloragaceae). Limnology and Oceanography 15:1-5.

_____. 1975. A treatise on limnology. III. Limnological Botany. Wiley, New York, New York, USA.

Jones, H. E. 1971. Comparative studies of plant growth and distribution in relation to waterlogging. III. The response of *Erica cinerea* L. to waterlogging in peat soils of differing iron content. Journal of Ecology 59:583-591.

Jones, R.. 1975. Comparative studies of plant growth and distribution in relationship to waterlogging. VIII. The uptake of phosphorus by dune and dune slack plants. Journal of Ecology 63:109-116.

Jupp, B. P., and D. H. N. Spence. 1977. Limitations on macrophytes in a eutrophic lake, Loch Leven. I. Effects of phytoplankton. Journal of Ecology 65:175-186.

Kadono, Y. 1980. Photosynthetic carbon sources in some *Potamogeton* species. Botanical Magazine, Tokyo 93:185-194.

_____. 1982. Occurrence of aquatic macrophytes in relation to pH, alkalinity, $Ca^{++}$, $Cl^{-}$ and conductivity. Japanese Journal of Ecology 32:39-44.

Keast, A. 1984. The introduced aquatic macrophyte, *Myriophyllum spicatum*, as habitat for fish and their invertebrate prey. Canadian Journal of Zoology 62:1289-1303.

Kemp, W. M., W. R. Boynton, R. R. Twilley, J. C. Stevenson, and J. Means. 1983. The decline of submerged vascular plants in upper Chesapeake Bay: summary of results concerning possible causes. Marine Science Technology 17:78-89.

Killgore, K. J. 1979. The ecological relationships of *Hydrilla verticillata* Royle in Lake Conroe, Texas. M.S. Thesis. Sam Houston State University, Huntsville, Texas, USA.

Kimball, K. D., and S. F. Kimball. 1977. Seasonal phytoplankton variations in the shallow Puhlavi Mordab, Iran. Hydrobiologia 55:49-53.

Klaine, S. J., and C. H. Ward. 1984. Environmental and chemical control of vegetative dormant bud production in *Hydrilla verticillata*. Annals of Botany 53:502-514.

Kollman, A. L., and M. K. Wali. 1976. Intraseasonal variations in environmental and productivity relations of *Potamogeton pectinatus* communities. Archiv fur Hydrobiologie 50:439-472.

Krapu, G. L., and G. A. Swanson. 1975. Some nutritional aspects of prairie nesting pintails. Journal of Wildlife Management 39:156-162.

Kullberg, R. G. 1974. Distribution of aquatic macrophytes related to papermill effluents in a southern Michigan stream. American Midland Naturalist 91:271-281.

Kulshreshtha, M., and B. Gopal. 1983. Allelopathic influence of *Hydrilla verticillata* (L.F.) Royle on the distribution of *Ceratophyllum* species. Aquatic Botany 16:207-209.

Lagler, K. F., J. E. Bardaih, and R. R. Miller. 1962. Ichthyology. Wiley. New York, New York, USA.

Lamberti, G. A., and J. W. Moore. 1984. Aquatic insects as primary consumers. Pages 164-195 in V. H. Resh and D. M. Rosenberg, editors. The ecology of aquatic insects. Praeger Publishers.

Langeland, K. A., and D. L. Sutton. 1980. Regrowth of hydrilla from axillary buds. Journal of Aquatic Plant Management 18:27-29.

Laughlin, D. R., and E. E. Werner. 1980. Resource partitioning in two coexisting sunfish: pumpkin-seed (*Lepomis gibbosus*) and northern longear sunfish (*Lepomis megalotis peltastes*). Canadian Journal of Fisheries and Aquatic Sciences 37:1411-1420.

Malmquist, B., L. M. Nilsson, and B. S. Svensson. 1978. Dynamics of detritus in a small stream in southern Sweden and its influence on the distribution of the bottom animal communities. Oikos 31:3-16.

Martin, M. H. 1968. Conditions affecting the distribution of *Mercurialis perennis* L. in certain Cambridgeshire woodlands. Journal of Ecology 56:777-793.

Martin, R. G., and J. V. Shireman. 1976. A quantitative sampling method for hydrilla-inhabiting macroinvertebrates. J. Aquat. Plant Manage. 14:16-18.

Mayer, F. L., and J. B. Low. 1970. The effect of salinity on widgeongrass. Journal of Wildlife Management 34:658-661.

McGahee, C. F., and G. J. Davis. 1971. Photosynthesis and respiration in *Myriophyllum spicatum* L. as related to salinity. Limnology and Oceanography 16:826-829.

Melis, R. J. M. 1984. Tuberization and hormones. Zeitschrift fuer Pflanzenphysiologie 113:271-283.

Menzie, C. A. 1980. The chironomid (Insecta: Diptera) and other fauna of a *Myriophyllum spicatum* L. plant bed in the Lower Hudson River. Estuaries 3:38-54.

Miller, J. L., L. A. Garrard, and W. T. Haller. 1976. Some characteristics of hydrilla tubers taken from Lake Ocklawaha during drawdown. Journal of Aquatic Plant Management 14:29-31.

Minshall, G. W. 1967. Role of allochthonous detritus in the trophic structure of a woodland spring brook community. Ecology 48:139-149.

_____, 1984. Aquatic insect-substratum relationships. Pages 358-400 in Vincent H. Resh and David M. Rosenberg, editors. The ecology of aquatic insects. Praeger Publishers.

Misra, R. D. 1938. Edaphic factors in the distribution of aquatic plants in the English Lakes. Journal of Ecology 26:411-451.

Mitra, E. 1964. Contributions to our knowledge of Indian fresh water plants. IV. On some aspects of morphological and anatomical studies of turions of *Hydrilla verticillata* (Linn. f.) Royle. Journal of Asiatic Society 6:17-27.

Mittelbach, G. G. 1981. Foraging efficiency and body size: a study of optimal diet and habitat use by bluegills. Ecology 62:1370-1386.

Morin, J. O., and K. D. Kimball. 1983. Relationship of macrophyte-mediated changes in the water column to periphyton composition and abundance. Freshwater Biology 13:403-404.

Moyle, J. B. 1945. Some chemical factors influencing the distribution of aquatic plants in Minnesota. American Midland Naturalist 34:402-420.

Mulligan, H. F., and A. Baranowski. 1969. Growth of phytoplankton and vascular aquatic plants at different nutrient levels. Verhandlungen des Internationalen Verein Limnologie 17:802-810.

Mulligan, H. F., A. Baranowski, and R. Johnson. 1976. Nitrogen and phosphorus fertilization of aquatic vascular plants and algae in replicated ponds. I. Initial response to fertilization. Hydrobiologia 48:109-116.

Nichols, S. A. 1974. Mechanical and habitat manipulation for aquatic plant management/a review of technique. Technical bulletin no. 76. Department of Natural Resources, Madison, Wisconsin, USA.

Orth, R. J., and J. V. Montfrans. 1984. Epiphyte-seagrass relationships with an emphasis on the role of micrograzing: a review. Aquatic Botany 18:43-69.

Ozimek, T. 1978. Effect of municipal sewage on the submerged macrophytes of a lake littoral. Ekologia Polska 26:3-39.

Paschal, J. E., Jr., D. R. Miller, N. C. Bartow, and V. Carter. 1982. Submersed aquatic vegetation in the tidal Potomac River and estuary of Maryland, Virginia, and the District of Columbia. Hydrologic data report, May 1978 to November, 1981. Open-file report 82-694. U.S. Department of the Interior, Geological Survey WRD, Reston, Virginia, USA.

Patten, B. C. 1956. Notes on the biology of *Myriophyllum spicatum* L. in a New Jersey Lake. Bulletin of the Torrey Botanical Club 83:5-18.

Pearsall, W. H. 1920. The aquatic vegetation of the English lakes. Journal of Ecology 8:163-199.

Pennak, R. W. 1953. Freshwater invertebrates of the United States. The Ronald Press Company, New York, New York, USA.

_____. 1971. Toward a classification of lotic habitats. Hydrobiologia 38:321-334.

Phillips, G. L., D. Eminson, and B. Moss. 1978. A mechanism to account for macrophyte decline in progressively eutrophicated freshwaters. Aquatic Botany 4:103-126.

Phillips, R. C., W. S. Grant, and C. P. McRoy. 1983. Reproductive strategies of eelgrass (*Zostera Marina* L.). Aquatic Botany 16:1-20.

Pieterse, A. H. 1981. *Hydrilla verticillata* - a review. Abstracts on Tropical Agriculture 7:9-34.

Pond, R. H. 1905. The relation of aquatic plants to the substratum. Report of the United States Fisheries Commission 19:483-526.

Ponnamperuma, F. N. 1972. The chemistry of submerged soils. Advanced Agronomy 24:29-96.

Pulich, W. M., Jr. 1982. Edaphic factors related to shoalgrass (*Halodule Wrightii Aschers.*) production. Botanica Marina 25:467-475.

Rabe, R., W. Nobel, and A. Kohler. 1982. Effects of sodium chloride on photosynthesis and some enzyme activities of *Potamogeton alpinus*. Aquatic Botany 14:159-165.

Ryan, J. B., D. N. Reimer, and S. J. Toth. 1972. Effects of fertilization on aquatic plants, water, and bottom sediments. Weed Science 20:482-486.

Sanderson, P. L., and W. Armstrong. 1980. Phytotoxins in periodically waterlogged forest soil. Journal of Soil Science 31:643-653.

Sand-Jensen, K. 1977. Effect of epiphytes on eelgrass photosynthesis. Aquatic Botany 3:55-63.

Sand-Jensen, K., and M. Søndergaard. 1981. Phytoplankton and epiphyte development and their shading effect on submerged macrophytes in lakes of different nutrient status. Internationale Revue dergesantum Hydrobiologie 66:529-552.

Sastroutomo, S. S. 1980. Dormancy and germination in axillary turions of *Hydrilla verticillata*. Botanical Magazine 93:265-273.

Savino, J. F., and R. A. Stein. 1982. Predator-prey interaction between largemouth bass and bluegills as influenced by simulated, submersed vegetation. Transactions of the American Fisheries Society 111:255-266.

Sculthorpe, C. D. 1967. The biology of aquatic vascular plants. Arnold, London, England.

Schwarzenbach, E. M. 1945. Kreuzungversuche an Hydrocharitaceae. Archiv der Julius Klaus-Stiflung fuer vererbungsferschung 20:22-41.

Seddon, B. 1972. Aquatic macrophytes as limnological indicators. Freshwater Biology 2:107-130.

Serie, J. R., and G. A. Swanson. 1976. Feeding ecology of breeding gadwalls on saline wetlands. Journal of Wildlife Management 40:69-81.

Singh, S. B., and S. Sahai. 1977. Dry matter production of submerged macrophytes of Jalwania pond of Gorakhpur, India. Geobios 42:75-76.

Smart, R. M., and J. W. Barko. 1985a in press. Laboratory culture of submersed freshwater macrophytes on natural sediments. Aquatic Botany.

Smart, R. M., and J. W. Barko. 1985b in press. Effects of water chemistry on aquatic plants: growth and distribution of *Myriophyllum spicatum* L. in relation to water chemistry. Technical report. Waterways Experiment Station, Vicksburg, Mississippi, USA.

Song, T. S. 1984. Phytochrome. Pages 354-379 in N. B. Wilkins, editor. Advanced plant physiology. Pitman Publishing Company, Marshfield, Massachusetts.

Spence, D. H. N. 1967. Factors controlling the distribution of freshwater macrophytes, with particular reference to Scottish lochs. Journal of Ecology 55:147-170.

Spence, D. H. N., and J. Chrystal. 1970a. Photosynthesis and zonation of freshwater macrophytes. I. Depth distribution and shade tolerance. New Phytologist 69:205-215.

_____. 1970b. Photosynthesis and zonation of freshwater macrophytes. II. Adaptability of species of deep and shallow water. New Phytologist 69:217-227.

Spencer, D. F., and L. W. J. Anderson. 1985. Effects of photoperiod on biomass and tuber production by two strains of *Hydrilla verticillata* (L.F.) Royle. Weed Sci. Soc. Amer. Abstracts.

Steward, K. K. 1969. Effects of growth regulators and herbicides on germination of *Hydrilla* turions. Weed Science 17:299-301.

Steward, K. K. 1984. Growth of *Hydrilla (Hydrilla verticillata)* in hydrosoils of different composition. Weed Science 32:371-375.

Steward, K. K., and T. K. Van. 1984. Comparative studies of monoecious and dioecious hydrilla biotypes. Florida Science 47:5.

Steward, K. K., T. K. Van, V. Carter, and A. H. Pieterse. 1984. *Hydrilla* invades Washington, D.C. and the Potomac. American Journal of Botany 71:162-163.

Swarbrick, J. T., C. M. Finlayson, and A. J. Cauldwell. 1982. The biology and control of *Hydrilla verticilla* (L.f.) Royle. Special Publication No. 16.

Swindale, D. N., and J. T. Curtis. 1957. Phyto-sociology of the larger submerged plants in Wisconsin Lakes. Ecology 38:397-407.

Titus, J. E., and M. S. Adams. 1979. Coexistence and the comparative light relations of the submersed macrophytes *Myriophyllum spicatum* L. and *Vallisneria americana* Michx. Oecologia 40:273-286.

Twilley, R. R, W. M. Kemp, K. W. Staver, and J. C. Stevenson. 1985. Nutrient enrichment of estuarine submersed vascular plant communities. I. Algal growth and effects on production of plants and associated communities. Marine Ecology Progress Series (in press).

Van, T. K. 1978. Some aspects of competitive biology of *Hydrilla*. Pages 117-126. Proceedings of 5th EWRS international symposium on aquatic weeds. Amsterdam, Netherlands.

Van, T. K., W. T. Haller, and G. Bowes. 1976. Comparison of the photosynthetic characteristics of three submerged aquatic plants. Plant Physiology 58:761-768.

Van, T. K., W. T. Haller, and L. A. Garrard. 1978. The effect of daylength and temperature on hydrilla growth and tuber production. Journal of Aquatic Plant Management 16:57-59.

Verhoeven, T. A. 1975. *Ruppia*-communities in the Camargue, France. Distribution and structure in relation to salinity and salinity fluctuations. Aquatic Botany 1:217-241.

Verhoeven, J. T. A., and W. Vierssen. 1978. Distribution and structure of communities structure in relation to salinity and salinity fluctuations. Aquatic Botany 1:217-241.

Waisel, Y., M. Agami, and Z. Shapira. 1982. Uptake and transport of $^{86}Rb$, $^{32}P$, $^{36}Cl$, and $^{22}Na$ by four submerged hydrophytes. Aquatic Botany 13:179-186.

Wallace, J. B., J. R. Webster, and W. R. Woodall. 1977. The role of filter feeders in flowing waters. Archiv fur Hydrobiologie 79:506-532.

Watkins, C. E., J. V. Shireman, and W. T. Haller. 1983. The influence of aquatic vegetation upon zooplankton and benthic macroinvertebrates in Orange Lake, Florida. J. Aquat. Plant Manage. 21:78-83.

Westlake, D. F. 1975. Macrophytes. Pages 106-128 in B. A. Whitton, editor. River ecology. University of California Press, Berkeley and Los Angeles, California, USA.

_____. 1981. Temporal changes in aquatic macrophytes and their environment. Dynamique de Populations et Qualite' de L'Eau 110-138.

Wetzel, R. G. 1975. Primary production. Pages 230-247 in B. A. Whitton editor. River ecology. University of California Press, Berkeley and Los Angeles, California, USA.

_____. 1983. Limnology. W. B. Saunders, Philadelphia, Pennsylvania, USA.

Wiley, M. J., R. W. Gorden, S. W. Waite, and T. Powless. 1984. The relationship between aquatic macrophytes and sport fish production in Illinois Ponds: A simple model. North American Journal of Fisheries Management 4:111-119.

Yeo, R. R, R. H. Falk, and J. R. Thurston. 1984. The morphology of *Hydrilla* (*Hydrilla verticillata*) (L.f.) Royle. Journal of Aquatic Plant Management 22:1-17.

Yoshida, T. 1975. Microbial metabolism of flooded soils. Pages 83-122 in E. A. Paul and A. D. Melaren, editors. Soil biochemistry. Dekker, New York, USA.

# CHAPTER III: BIOLOGICAL CONTROL

## Overview

Biological control refers to the use of living organisms to suppress the population of a plant pest. It includes the augmentation of endemic enemies and release of exotic natural enemies. Ideally, a biocontrol agent is specific, attacking only the target pest, and maintains population levels adequate to suppress the pest population below the problem level.

### Insects

Historically biological control efforts on *Hydrilla* dealt mainly with the classical approach. The term "classical approach" is used to describe the reassociation of a foreign pest with its natural enemies from its country of origin. Not long after *Hydrilla* had been identified in the US and its potential as a pest recognized, interest in search for insects to control *Hydrilla* began to increase. Foreign scientists were contracted to conduct searches and US scientists made brief overseas trips in search of natural insect enemies on *Hydrilla* (Rao 1969, Rao and Sankaran 1974, Varghese and Singh 1976, Pemberton 1980). These early efforts met with little success.

In 1978, the Aquatic Plant Control Research Program (APCRP)* funded the US Department of Agriculture (USDA) to conduct a domestic survey for endemic enemies of *Hydrilla* (Balciunas and Minno 1985). In 1981, the APCRP initiated overseas searches for insects on *Hydrilla* in Southeast Asia, India, and Australia (Balciunas 1985). Thus far, the searchers have collected over 45 insect species that feed on *Hydrilla* and seem to be host specific.

### Fish

The use of fish as biological controls for *Hydrilla* has received a great deal of attention. The grass carp (*Ctenopharyngodon idella*), also called the white amur, has demonstrated tremendous potential for biocontrol of *Hydrilla*. This large herbivorous fish consumes enormous amounts of aquatic vegetation. While the grass carp will feed on almost any vegetation, including terrestrial vegetation that comes in contact with water, *Hydrilla* is a preferred food.

---

* APCRP is sponsored by the Civil Works Directorate of the Office, Chief of Engineers, and is managed by the US Army Engineer Waterways Experiment Station.

Grass carp are apparently effective in keeping small enclosed aquatic systems free of *Hydrilla*. The fish were once considered to be unable to breed successfully in US waters, but this assumption has since been proven incorrect (Pierce 1983). The fish *Tilapia zillii* (Gervais) also consumes *Hydrilla* (Legner 1979), but this fish is much smaller and does not damage *Hydrilla* nearly as much as the grass carp.

Two species of fish from South America, *Metynnis roosevelt* and *Mylossoma argenteum*, known as silver dollar fish, can control a variety of aquatic plants. Although both species are quite small (approximately 13 cm total length), when they travel in schools they often consume large quantities of plant material. These species are mowers: they bite the plants at the base close to the roots. When the vegetation floats to the surface, the silver dollar either graze on it or leave it to decay. When stocked at rates of 1,200 to 2,500/hectare, these little fish have been known to control dense growths of vegetation.

These fish require warm water, usually 16° to 35°C. Temperatures below 16°C are fatal. In South America, silver dollar fish are sought after as food. They do resemble the piranha and are often mistaken for that dreaded carnivore (National Academy of Science 1976). Since these fish require warm water and are not native to the US, they should not be considered as macrophyte control agents for *Hydrilla* in the Potomac River.

Microorganisms

*Fusarium roseum* Culmorum, an exotic fungal plant pathogen, has been shown to have potential as a biocontrol agent for *Hydrilla* (Charudattan et al. 1980). However, the lack of specificity in preemergent seed viability tests and the difficulty in demonstrating efficacy on a large scale resulted in the discontinuation of development of the fungus for biocontrol purposes (Charudattan et al. 1983).

APCRP researchers are working on new approaches to control *Hydrilla* with microorganisms. Microorganisms that produce metabolites injurious to the plant are selected from the natural microflora of *Hydrilla*. These organisms are then conditioned for maximum production of the metabolites and reintroduced to the plants. The possibility of genetically engineering a microbial control agent is also being investigated, but is still in the planning stages.

Other potential biocontrol agents

The snail, *Marisa cornuarietis*, consumes *Hydrilla* and has been considered for use as a biological control agent. However, large numbers are necessary to achieve control, and *Marisa* is not completely specific. It feeds on young rice plants and other plant species (Blackburn et al. 1971).

The manatee, *Trichechus manatus* L., consumes enormous amounts of aquatic vegetation, including *Hydrilla* (Campell and Irvine 1977). However, the manatee is a timid, fragile animal that is restricted to the warm spring waters of Florida and is easily injured by boat traffic, making it impractical for use in management programs.

## Control with Insects

Domestic survey

A survey of the macroinvertebrates associated with *Hydrilla* in the US was conducted between July 1978 and August 1980 (Balciunas and Minno 1985). A total of 285 collections at 76 sites resulted in 59,010 macroinvertebrate specimens. Of these, 17,358 (29.4 percent) were insects representing 191 species. The insects that caused the most damage were the larvae of aquatic moths with *Parapoynx diminutalis* and *Synclita obliteralis* being the most common. *Synelita obliteralis* is not host specific and therefore cannot be considered as a biocontrol agent for *Hydrilla*. *Parapoynx diminutalis*, an Asiatic species, was the only insect showing a preference for *Hydrilla* in the field. *Parapoynx diminutalis* was found in the US in 1975 (Del Fosse et al. 1976a). The moth, probably cointroduced with *Hydrilla*, is now widespread in Florida and already negatively impacting *Hydrilla* at some locations (Balciunas and Habeck 1981). Numerous midges (Diptera:Chironomidae) and leptocerid caddisflies (Trichoptera:Leptoceridae) were frequently found on *Hydrilla* but only occasionally were observed to cause damage.

Overseas surveys

Thus far, overseas searches in Southeast Asia, India, and Australia have resulted in the identification of at least 45 insect species that feed on *Hydrilla* and appear to be host specific (Balciunas 1985).

Almost 20 weevil species, mostly in the genus *Bagous*, were collected. These weevils are of special interest since they usually are host specific and have short life cycles, thus making them ideal biocontrol candidates. Of

special interest are the Indian weevils whose larvae feed on *Hydrilla* tubers. These were collected in Bangalore, India, in 1982 and were brought back to US quarantine facilities for further evaluation. Host testing in quarantine has demonstrated that one *Bagous* sp. is specific to *Hydrilla* and permission to release this species in the US may be sought (Dr. Gary Buckingham, personal communication*).

Another insect group showing good potential as biological control agents are the ephydrid flies, especially in the genus *Hydrellia*. Several species were tested in Pakistan as potential biological control agents of *Hydrilla*, and *Hydrellia pakistanae* was found to be both effective and host specific (Baloch et al. 1980). Approval has been granted to evaluate *H. pakistanae* in US quarantine facilities as a potential biocontrol agent for *Hydrilla* (Dr. Gary Buckingham, personal communication*).

Aquatic moths in the family Pyralidae have demonstrated potential to control *Hydrilla* (Rao 1969, Rao and Sankaran 1974, Ghani 1976, and Varghese and Singh 1976). *Parapoynx diminutalis* is common in South Asia and widespread throughout Southeast Asia. A similar species, *P. dicentra*, was found in northern Australia. These *Parapoynx* sp. cause the most easily observed damage to *Hydrilla* and, when present in large numbers, completely defoliate the plant.

*Parapoynx diminutalis* was also found to impact *Hydrilla* in Panama and was brought into US quarantine facilities for evaluation (Balciunas and Center 1981, Buckingham and Bennett 1984). Buckingham and Bennett concluded that *P. diminutalis* was polyphagous (Table 1). Of the aquatic plant species that occur in the Potomac River, *P. diminutalis* would pose a real threat to *Ceratophyllum demersum* and *Vallisneria americana* in or near the site of release (based on laboratory studies). It might also damage *Zannichellia palustris*, *Myriophyllum spicatum*, and the *Najas* sp. in the vicinity of the release but damage would probably be minimal. Otherwise, the insect would be a desirable biocontrol in the temperate climate of Virginia because it cannot overwinter outside the tropics and would not pose a threat to aquatic plants outside the site of release. Annual releases on infested areas would be required.

---

* Dr. Gary Buckingham, US Dept of Agriculture, Pest Control Research Unit, Gainesville, Fla., 1985.

**Table 1**
**Summary of Greenhouse No-Choice Development Tests with Multiple Larvae of *Parapoynx diminutalis****

| *Test Plants* | *Common Name* | *Total Eggs Tested* | *Total Number of Replicates* | *Type of Tests*** | *% Adults Mean ± SD (Range)* |
|---|---|---|---|---|---|
| *Nymphaea* sp.† + *N. odorata*†† | Waterlily | 10 | 1 | C | 50 |
| *Hydrilla verticillata* | Hydrilla | 250 | 8 | A, B | 47.75 ± 21.7 (14-68) |
| *Cabomba pulcherrima* | Purple fanwort | 98 | 7 | B, C, G | 40.7 ± 31.9 (0-90) |
| *Hygrophila polysperma* | Hygrophila | 110 | 2 | C, E | 38.0 ± 45.2 (6-70) |
| *Egeria densa* | Egeria | 50 | 2 | B | 34.0 ± 25.4 (16-52) |
| *Vallisneria americana* | Watercelery | 160 | 13 | C, D | 27.3 ± 21.3 (0-70) |
| *Najas guadalupensis* | Southern naiad | 50 | 2 | B | 24.0 ± 33.9 (0-48) |
| *Ceratophyllum demersum* | Coontail | 70 | 7 | C | 22.9 ± 21.4 (0-84) |
| *Najas minor* + | Slender naiad | | | | |
| *N. guadalupensis*†† | Southern naiad | 50 | 2 | B | 20.0 ± 5.6 (16-24) |
| *Zanichellia palustris* + | Horned pondweed | | | | |
| *Eleocharis* sp.†† | Dwarf spikerush | 60 | 3 | D | 16.6 ± 28.8 (0-50) |
| *Eleocharis* sp. | Dwarf spikerush | 100 | 2 | A | 13.0 ± 1.4 (12-14) |
| *Myriophyllum spicatum* | Eurasian watermilfoil | 120 | 4 | A, C | 7.5 ± 8.6 (0-20) |
| *Mayaca fluvialis* | Bogmoss | 100 | 2 | A | 5.0 ± 4.2 (2-8) |
| *Ruppia maritima* | Widegeongrass | 150 | 4 | A, B | 4.5 ± 5.2 (0-12) |
| *Polygonum* sp. | Smartweed | 50 | 1 | A | 4.0 |
| *Myriophyllum heterophyllum* | Broadleaf watermilfoil | 150 | 4 | A, B | 4.0 ± 4.6 (0-8) |
| *Nymphaea odorata* | Fragrant waterlily | 250 | 6 | A, B | 0.7 ± 1.6 (0-4) |
| *Utricularia foliosa* | Bladderwort | 50 | 2 | B | 0 |
| *Utricularia biflora* | Bladderwort | 50 | 2 | B | 0 |
| *Proserpinaca palustris* | Mermaidweed | 20 | 2 | C | 0 |
| *Oryza sativa* L. 'Saturn' | Saturn rice | 220 | 3 | C, G | 0 |
| *Nuphar sagittifolium* | Spatterdock | 210 | 3 | C, G | 0 |
| *Nitella* sp. | Nitella | 20 | 2 | C | 0 |
| *Salvinia rotundifolia* | Common salvinia | 100 | 2 | A | 0 |
| *Lemna minor* | Common duckweed | 100 | 2 | A | 0 |
| *Bacopa caroliniana* | Bacopa | 179 | 4 | A, H | 0 |
| *Azolla caroliniana* | Waterfern | 150 | 4 | A, B | 0 |
| *Nuphar advena* | Spatterdock | 170 | 4 | A, D | 0 |
| *Nymphoides aquaticum* | Bananalily | 200 | 4 | A | 0 |
| *Pistia stratiotes* | Waterlettuce | 150 | 4 | A, B | 0 |
| *Sagittaria subulata* complex | Arrowhead | 130 | 7 | C, D | 0 |
| *Sagittaria isoeteformis* | Arrowhead | 60 | 3 | D | 0 |
| *Potamogeton illinoensis* | Illinois pondweed | 120 | 6 | D | 0 |
| *Marsilea* sp. | Marsilea | 100 | 2 | A | 0 |
| *Isoetes* sp. | Isoetes | 100 | 2 | A | 0 |
| *Echinodorus* sp.† | Burhead | 120 | 4 | A,C | 0 |
| *Nasturtium officinale* | Watercress | 100 | 2 | A | 0 |
| *Limnobium spongia* | Frogbit | 70 | 4 | B,C | 0 |

* All tests were initiated with eggs that had well-developed larvae visible inside; replicates were initiated when larvae were available from November 1980 through May 1981.

** A - 50 eggs/0.95-*l* jar; B - 25 eggs/0.95-*l* jar; C - 10 eggs/0.95-*l* jar; D - 20 eggs/3.79-*l* jar; E - 100 eggs/0.95-*l* jar; F - 200 eggs/3.79-*l* jar; G - 8 eggs/0.95-*l* jar; H - 29 eggs/0.95-*l* jar; I - 5 eggs/0.95-*l* jar

† Ornamental species from aquatic plant dealer.

†† Test initiated with the first species, but the second species was substituted when the first was no longer available.

Note: This table was taken from Buckingham and Bennett (1984).

Host specificity and efficacy studies on the more promising insects found in Australia began in 1985. The insects that prove to be host specific will be brought into quarantine in the US in anticipation of eventual release.

## Control with Fish

The grass carp or white amur (*Ctenopharyngodon idella*), native to the Amur Basin in eastern China (Berg 1949), is known from 50 countries worldwide including Canada, Egypt, France, India, Iran, United States, Nigeria, Sweden, and Yugoslavia. Grass carp were brought to the United States in 1963 for study by the Arkansas Game and Fish Commission. Since then they have been introduced into 35 states ranging from Florida, Texas, and California in the south to Vermont, Michigan, Minnesota, and Oregon in the north. Although they are used mainly to control plants in the United States, this species is an important source of protein in European and eastern countries.

While adult grass carp are herbivorous, their larvae commonly eat phytoplankton and zooplankton (Linchevskaya 1966, Rozmanova 1966). When they are about 2 cm long, grass carp begin to eat macrophytes and usually feed on plants for the rest of their lives. However, there are reports of subadults eating animal matter; Singh et al. (1976) found that 7- to 13-cm grass carp avidly ate common carp hatchlings while 20- to 25-cm specimens refused them. In devegetated ponds, juvenile grass carp apparently resorted to insects for food (Kilgen and Smitherman 1971, 1973; Forester and Avault 1978), although Colle et al. (1978) found only trace amounts of invertebrates in 6- to 22-cm grass carp.

Adult grass carp preferentially feed on succulent macrophytes such as *Nitella* sp., *Najas* sp., *Hydrilla*, *Elodea canadensis*, *Pithophora* sp., and *Ceratophyllum demersum* (Table 2). They will also eat *Myriophyllum* sp., *Bacopa* sp., *Egeria densa*, *Spirogyra* sp., *Utricularia* spp., *Cabomba* spp., and *Brasenia schreberi*. Plants that grass carp eat but only when preferred species are absent include *Vallisneria* spp., *Thypha* spp. *Myriophyllum brasilience*, *Carex* spp., *Scirpus* spp. and *Phragmites* spp. (Nall and Schardt 1978). When food supplies are low, adult grass carp have been observed eating terrestrial plants along the shore; however, they do not feed on other fish or fish eggs when aquatic plants are absent.

Table 2

Grass Carp Feeding Preferences for Aquatic Plants Found in the Potomac River*

| Plant Species | Preference** |
|---|---|
| *Ceratophyllum demersum* | RC |
| *Elodea canadensis* | RC |
| *Heteranthera dubia* | ND |
| *Hydrilla verticillata* | RC |
| *Myriophyllum spicatum* | RC |
| *Najas* sp. | RC |
| *Nitella flexilis* | RC |
| *Potamogeton crispus* | RC |
| *P. pectinatus* | RC |
| *P. perfoliatus* | RC |
| *P. pusillus* | RC |
| *Ruppia maritima* | ND |
| *Vallisneria americana* | PC |
| *Zannichellia palustris* | PC |

* From Bailey (1972) and Nall and Schardt (1978).

** Entries in this column are defined as follows: RC = readily consumed; ND = no data; PC = partially consumed.

Grass carp have a horny pad on the roof of their mouth and pharyngeal teeth that consist of a double row of finely serrated structures. They feed by grasping plants between the horny pad and the pharyngeal teeth and shaking from side to side until the stems break. Unlike the common carp, which pulls and uproots vegetation, the grass carp does not muddy the water as it feeds. Plant material is macerated by the action of the pharyngeal teeth against each other and the horny pad. As fish grow older, the pharyngeal teeth increase in size and grow further apart, thus allowing grass carp to eat very fibrous material. A mature fish can eat a cattail by cutting it at the base below the water level and consuming the entire plant from base to tip.

Grass carp have been reported to compete with crayfish for aquatic plants in small ponds (Forester and Avault 1978) and to reduce the food base for herbivorous overwintering waterfowl (Gasaway and Drda 1977). Because they are selective feeders, grass carp may cause an increase in unpalatable plants at the expense of preferred species (Vinogradov and Zolotova 1974).

Grass carp are known to be voracious feeders; at optimal temperatures, they can consume from 2/3 to 3 times their own weight in a single day. High feeding rates are the result of incomplete digestion and rapid passage of poorly macerated material through their short intestine. Consumption rates are slowed by increased salinity, decreased oxygen concentrations, physical disturbance, and abrupt changes in temperature. The fish reportedly do not feed at all when water temperatures are below 14°C; between 14° and 16°C, they are sluggish and feed selectively. At water temperatures above 20°C, they can become ravenous and consume preferred and nonpreferred plants. Feeding rates remain relatively constant from 23° to 26°C and then begin to decline. The grass carp tolerates a broader range of water temperatures than other phytophagous fish with potential for weed control (Prowse 1969).

The rapid elimination of macrophytes and abrupt influx of nutrients in the grass carp's feces can result in bluegreen algae blooms (Alikunhi and Sukumaran 1964). Several workers have reported local changes in phytoplankton species composition and abundance (Gasaway 1977a, b), although Fry and Osborne (1980) observed no changes in zooplankton that could be directly attributable to grass carp introduction. Since invertebrate density and diversity are positively correlated with macroinvertebrate densities, total elimination of plants has negative effects on aquatic systems. In other studies, no significant changes in macroinvertebrates have been noted with introduction of grass

carp because not all plants were eliminated (Rottmann 1976, Rottmann and Anderson 1976, Crisman and Kooijman 1980).

There have been numerous investigations concerning tolerance of fry and adults to changes in water temperature, dissolved oxygen, pH, salinity, turbidity, alkalinity, ammonia, chlorine, and sulfide (Singh et al. 1967, Opuszynski 1979, Custer et al. 1978). These fish cease feeding when dissolved oxygen levels reach about 2.5 mg/ℓ. At salinities greater than 30 percent seawater, mortalities occur; growth rates slow appreciably at lower salinities.

A monosex (all female) grass carp population can be produced by artificial gynogenesis, a process where sperms are irradiated to destroy their capacity to produce males. The resulting females are fed sex-reversal hormones prior to formation of sex organs. These sex-reversed females (actually males) carry chromosomes capable of producing females; when they are paired with normal females, the offspring are all female. If monosex (all female) fish are stocked, natural reproduction occurs only if a male from another source joins the population.

Grass carp can be crossed with male bighead carp (*Hypophtalmichtys nobilis*), yielding matroclinous (showing characteristics from the maternal side) and intermediate offspring that are triploid. The triploid, which is sterile, has been found to be as effective a macrophyte control agent as the diploid. Grass carp have been crossed with female silver carp (*Hypophthalmichtys molitrix*), goldfish (*Carassius auratus*), eastern bream (*Abramis brama orientalis*), black bream (*Megalobrama terminalis*), white bream (*Parabramis pekinesis*), and rohu carp (*Labeo rohita*) as well as other Chinese carps (Smith and Shireman 1983).

Grass carp usually spawn their pelagic eggs in the primary channels of large rivers. Conditions that give rise to reproduction are increases in water levels, temperatures above 17°C, and current velocities that are more than 0.6 m/sec. Spawning grounds usually occur in an area immediately downstream of a tributary, island, or other geologic feature that causes strong vertical mixing and has rock, gravel, or sand substrate. Although the reproductive requirements of this species are fairly specific, immature grass carp have been found in large rivers in the United States, presumably the result of natural reproduction. However, there have been no reports to date

of large numbers of grass carp establishing themselves naturally in the United States.

Water-quality changes resulting from using grass carp introduction include increased turbidity and potassium in Indiana ponds (Lembi and Ritenour 1977), and decreased pH and increased Kjeldahl nitrogen in a Florida lake (Kobylinski et al. 1980). Lembi and Ritenour (1977) and Lembi et al. (1978) reported that grass carp can cause decreased dissolved oxygen and increased carbon dioxide in small ponds. Beach et al. (1976, 1977) and Gasaway (1977a,b) documented increases in nitrate and chlorophyll levels in four Florida ponds after stocking with grass carp. In one study, there were no water-quality changes (Fry and Osborne 1980), and Mitzner (1975) and Michewicz et al. (1972) reported that grass carp effects were related primarily to size of water body.

At Lake Conway, Florida, more than 8000 grass carp (0.25 to 0.61 g) were stocked at the rate of 3 to 5/acre. As a result, three of the four most common plants, *Hydrilla*, *Nitella* sp., and *Potamogeton* sp., were reduced by 90 percent in 18 months. *Vallisneria* sp., not readily consumed by grass carp, was relatively unaffected (Miller and King 1984). Reported stocking rates for grass carp vary from less than 1.0 lb/acre (Osborne and Sassic 1979), to 50 lb/acre (Bailey 1972). Stocking rates varied depending upon temperatures, total area involved, water quality, level of plant infestations, nature of the problem, and objectives of the study. A stocking rate model and a review of specific stocking rates that proved successful under various conditions appear in Miller and Decell (1984).

Grass carp are best employed as macrophyte control agents in ponds, lakes, or other freshwater sites where they can be easily contained. Since nonreproducing fish are usually stocked, grass carp can be reintroduced after several years if deemed appropriate. The grass carp presents a viable alternative to other commonly used plant-control measures. Numerous workers have reported that they last longer and cost less to use than other measures (Bailey 1972, Beach 1973, and other references cited in Smith and Shireman 1983). The use of grass carp in conjunction with other control measures has been proposed and tested. For example, the combination of grass carp and mottled water hyacinth weevil (*Neochetina eichhorniae*) retarded water hyacinth more than either organism by itself (Del Fosse et al. 1976b). In India manual

methods of plant control have been used in conjunction with grass carp (Singh 1976).

## Control with Pathogens

Since no successful biocontrol agents have yet been generated by conventional approaches, two new approaches are being evaluated by the APCRP to generate pathogens of *Hydrilla*. These approaches are (1) enhancement of lytic-enzyme-producing microorganisms and (2) genetic engineering.

### Lytic-enzyme-producing microorganisms

Among the natural microflora of aquatic plants are saprophytes and weak pathogens that cause the breakdown of plant tissues during senescence late in the growing season. These organisms normally produce lytic enzymes specific for certain plant tissues (i.e., cellulose and pectin). If such mircoorganisms can be induced to increase their production of lytic enzymes, they may become capable of attacking plant tissues at any time during the growing season.

A search was conducted for lytic-enzyme-producing microorganisms on dioecious *Hydrilla* of the southeast and western US. Two hundred sixteen microorganisms were isolated from *Hydrilla* and screened for production of the lytic enzymes cellulase and pectinase (Pennington 1985). Cellulase digests the cellulose in plant cell walls and pectinase digests pectin, a cementitious material that holds plant cells together. Twenty-two of the isolates produced lytic enzymes. These isolates were successively subcultured on restrictive media in the laboratory to enhance enzyme production. The enzyme-enhanced isolates were introduced to *Hydrilla* sprigs in test tubes to determine whether they could damage the plant. Six of the isolates produced extensive damage to *Hydrilla*. These isolates are being evaluated in aquarium and greenhouse tank studies in order to scale-up the research to a field situation. Host-specificity studies will not be conducted unless or until greenhouse studies are successfully completed. Therefore, the potential threat to other plant species within the Potomac River has not been determined. The results are promising thus far. Field studies will be initiated in 1986.

### Genetic engineering

Application of genetic engineering technology is another possible approach to the development of biocontrol agents of *Hydrilla*. Microorganisms

have already been engineered to solve specific biological problems. For example, genetically engineered microorganisms are now mass producing viral inhibitors (interferon), insulin, and other commercially valuable pharmaceuticals that were previously available in extremely limited quantities.

In 1983 three experts involved in different aspects of genetic engineering technology and research met at the Waterways Experiment Station with members of the APCRP. The feasibility of developing a pathogen for biocontrol of submersed aquatic plants was discussed. The consensus of scientists at the meeting was that genetic engineering technology is sufficiently advanced to develop a safe, effective biocontrol agent for submersed aquatic plants. The crucial prerequisite is identifying microorganisms that are already host specific to the target plant. Once host-specific microorganisms are found, the engineering process should require 5 to 7 years. The microorganisms produced would then require efficacy testing, formulation of a usable product for field application, and reaffirmation of host specificity. These tests are the same as those required for the release of naturally occurring pathogens.

The APCRP will initiate a domestic search for host-specific microorganisms in fiscal year 1986 with plans to develop any candidates found by application of genetic engineering technology.

## Conclusions and Recommendations

### Insects

Overseas searches for insect biocontrol agents have resulted in the introduction of two potential biocontrol agents into US quarantine facilities (*Bagous* weevil and *Hydrellia* fly). It is expected that several more insects will be cleared for quarantine studies in the near future as a result of current research in Australia. Approval for release of one of the insects now in quarantine could be gained by 1986. When approval is granted, the insect should be evaluated on the monoecious variation of *Hydrilla* from the Potomac River. It is recommended that this research be included in any contingency plans for the Potomac River effort.

*Hydrilla* infestations in Asia, Australia, and Panama were found to be heavily damaged by *Parapoynx diminutalis* (pyralid moth). Quarantine studies demonstrated this species to be efficacious on *Hydrilla* in the US. Accidental releases have resulted in its establishment in the southeast US. The colder

winter water temperatures in the northeast will prevent its natural migration to the Potomac River.

It is recommended that *P. diminutalis* be field tested on monoecious *Hydrilla* in the Potomac River. Initially, *P. diminutalis* could be evaluated in the laboratory in 1985 for its potential as a biocontrol agent on monoecious hydrilla. Mass-rearing capabilities would be addressed and state permits obtained the following year (1986). A large-scale operations management test could be initiated in the Potomac River in 1987, assuming that the insect demonstrates potential in the laboratory and rearing methods are developed.

Grass carp

There are no known major detrimental environmental effects associated with the proper use of the grass carp to control macrophytes. When stocked at rates commensurate with the problem, native fish, waterfowl, reptile, and amphibian populations are unaffected. Water quality and macroinvertebrates are not directly affected, although in some cases the numbers of blue-green algae increase following removal of larger plants. The presence of grass carp will probably not upset existing relationships among aquatic organisms inhabiting the area although their effects on macroinvertebrate populations associated with aquatic vegetation should be evaluated.

If grass carp are used in the Potomac River, there should be no concern over containment since the fish will be blocked by saline waters in the bay and by the falls in the upper reach of the river. The triploid (sterile) form is a logical choice since it is nonreproductive and recent reports indicate that it consumes plants as well as the diploid.

It is recommended that any control plans for the river should be coupled with an analysis of the overall benefits of macrophytes to the aquatic biota. The grass carp will not have more pronounced indirect negative effects on the aquatic system than other control measures. These fish have been used to successfully control macrophytes in this country for the last 20 years and could be effectively utilized in the Potomac River. However, it should be noted that grass carp cannot be expected to concentrate feeding on specific problem sites such as access channels to marinas.

Lytic-enzyme-producing microorganisms

The ongoing biocontrol effort with dioecious *Hydrilla* utilizing lytic-enzyme-producing microorganisms has met with a great deal of success. In

laboratory studies, several isolates have demonstrated effectiveness for the destruction of dioecious hydrilla. It is anticipated that candidate biocontrol agents will be field tested in 1986. The monoecious variation of *Hydrilla* for the Potomac River should be included in the evaluation process as soon as possible to verify efficacy by the candidate biocontrol agents.

It is recommended that the candidate lytic-enzyme-producing biocontrol agents be field tested on monoecious hydrilla in the Potomac River. The greenhouse studies will be completed in 1985 and a small-scale field study will be conducted in 1986. A large-scale operations management test could be initiated in the Potomac River in 1987, assuming the research progresses as planned.

Genetic engineering

The genetic engineering research is still in the planning stages and should not be included in the Potomac River effort.

## Contributors


| | |
|---|---|
| Dr. Joseph K. Balciunas | US Department of Agriculture<br>Aquatic Plant Management Laboratory<br>3205 SW College Avenue<br>Fort Lauderdale, FL 33314 |
| Dr. Ted D. Center | US Department of Agriculture<br>Aquatic Plant Management Laboratory<br>3205 SW College Avenue<br>Fort Lauderdale, FL 33314 |
| Dr. Andrew C. Miller | US Army Engineer Waterways Experiment Station<br>Environmental Laboratory<br>P.O. Box 631<br>ATTN: WESER-A<br>Vicksburg, MS 39180 |
| Mr. Edwin A. Theriot* | US Army Engineer Waterways Experiment Station<br>ATTN: WESER-W<br>P.O. Box 631<br>Vicksburg, MS 39180 |


---

* Mr. Theriot was primarily responsible for the development of this chapter and should be used as the point of contact regarding its content.

## References


Alikunhi, K. H., and K. K. Sukumaran. 1964. Preliminary observations on Chinese carps in India. Proceedings of the Indian Academy of Sciences 60B(3):171-189.

Bailey, W. M. 1972. Arkansas evaluation of the desirability of introducing the white amur (*Ctenopharyngodon idella*, Val.) for control of aquatic weeds. Presented at 102nd Annual Meeting of the American Fisheries Society and International Association of the Game and Fish Commission, Hot Springs, Ark. 59 pp (mimeo).

Balciunas, J. K. 1985 (In Preparation). Final report on the overseas surveys (1981-1983) for insects to control *Hydrilla*. Miscellaneous Paper. US Army Engineer Waterways Experiment Station, Vicksburg, Miss. 111 pp.

Balciunas, J. K., and T. D. Center. 1981. Preliminary host specificity tests of a Panamanian *Parapoynx* as a potential biological control agent for *Hydrilla*. Environ. Entomol. 10(4):462-467.

Balciunas, J. K., and D. H. Habeck. 1981. Recent range extension of a hydrilla-damaging moth, *Parapoynx diminutalis* (Lepidoptera:Pyralidae). The Florida Entomo. 64(1):195-196.

Balciunas, J. K., and M. K. Minno. 1985 (In Preparation). Quantitative survey of the insects and other macrofauna associated with *Hydrilla*. Miscellaneous Paper. US Army Engineer Waterways Experiment Station, Vicksburg, Miss. 223 pp.

Baloch, G. M., Sana-Ullah, and M. A. Ghani. 1980. Some promising insects for the biological control of *Hydrilla verticillata* in Pakistan. Tropical Pest Manag. 26(2):194-200.

Beach, M. L. 1973. A study to determine the feasibility of using Chinese grass carp (*Ctenopharyngodon idella* Val.) as a biological control agent for noxious aquatic vegetation. Annual Report of the Florida Department of Natural Resources. Tallahassee, Fla. 173 pp.

Beach, M. L., R. L. Lazor, and A. P. Burkhalter. 1977. Some aspects of the environmental impact of the white amur (*Ctenopharyngodon idella* Val.) in Florida, and its use for aquatic weed control. In: Freeman, T. E. (ed.). 1977 Proceedings of the 4th International Symposium on Biological Control of Weeds. Gainesville, Fla.: University of Florida, Institute of Food and Agricultural Sciences. pp 269-289.

Beach, M. L., W. W. Miley, II, J. Van Dyke, and D. M. Riley. 1976. The effect of the Chinese grass carp (*Ctenopharyngodon idella*) (Val.)) on the ecology of four Florida lakes, and its use for aquatic weed control. Final Report. Tallahassee, Fla.: Florida Department of Natural Resources. 246 pp.

Berg, L. S. 1949. Freshwater fishes of the USSR and adjacent countries. (Ryb persenykh vod SSSR i soprendel'nykh stran). Vol. II. Jerusalem: Israel Program for Scientific Translations. pp 134-135.

Blackburn, R. D., K. A. Boyer, and C. E. Timmer. 1971. Phytotoxicity of four formulations of the alkylanine salt of endothal on *Hydrilla verticillata* and fish. Hyacinth Control Journal 9(1):55-58.

Buckingham, G. R., and C. Bennett. 1984. Laboratory biology and host range studies of *Parapoynx diminutalis* (Lepidoptera:Pyralidae). Miscellaneous Paper A-84-2, US Army Engineer Waterways Experiment Station, Vicksburg, Miss., pp 173-193.

Campbell, H. W., and A. B. Irvine. 1977. Feeding ecology of the West Indian manatee (*Trichechus manatus* Linnaeur). Aquaculture 12(3):249-251.

Charudattan, R., T. E. Freeman, R. E. Cullen, and F. M. Hofmeister. 1980. Evaluation of *Fusarium roseum* Culmorum as a biological control for *Hydrilla verticillata*. Abstracts of papers presented at the V International Symposium on Biological Control of Weeds; July 1980, Brisbane. p 7.

Charudattan, R., T. E. Freeman, R. E. Cullen, and F. M. Hofmeister. 1983. Status of the fungii *Cercospora* and *Fusarium* for aquatic plant control. Proceedings, 17th Annual Meeting, Aquatic Plant Control Research Program; November 1982, Sacramento, Calif., Miscellaneous Paper A-83-3. US Army Engineer Waterways Experiment Station, Vicksburg, Miss., pp 115-120.

Colle, D. E., J. V. Shireman, and R. W. Rottmann. 1978. Food selection by grass carp fingerlings in a vegetated pond. Transactions of the American Fisheries Society 107(1):149-152.

Crisman, T. L., and F. M. Kooijman. 1980. Large-scale operations management test using the white amur at Lake Conway, Florida. Benthos. In: Decell, J. L. (Ed.). 1980 Proceedings 14th Annual Meeting, Aquatic Plant Control Research Planning and Operation Review. US Army Engineer Waterways Experiment Station, pp 298-304.

Custer, P. E., and F. D. Halverson, J. Malone, and C. Von Chong. 1978. The white amur as a biological control agent of aquatic weeds in the Panama Canal. Fisheries 3(5):2-9.

Del Fosse, Ernest S., B. David Perkins, and Kerry L. Steward. 1976a. A new U.S. record for *Parapoynx diminutalis* (Lepidoptera:Pyralidae), a possible biological control agent for *Hydrilla verticillata*. Florida Entomol. 59:19-20.

Del Fosse, E. S., D. L. Sutton, and B. D. Perkins. 1976b. Combination of the mottled waterhyacinth weevil and the white amur for biological control of waterhyacinth. Journal of Aquatic Plant Management 14:64-67.

Forester, T. S., and J. W. Avault, Jr. 1978. Effects of grass carp on freshwater red swamp crayfish in ponds. Transactions of the American Fisheries Society 107(1):156-160.

Fry, D. L., and J. A. Osborne. 1980. Zooplankton abundance and diversity in central Florida grass carp ponds. Hydrobiologia 68(2):145-155.

Gasaway, R. D. 1977a. The effects of grass carp on community structure in four Florida lakes. In: Florida Game and Fresh Water Fish Commission. 1977. The Grass Carp: A Special Research Report to the Governor and the Cabinet. Tallahasee, FL: Florida Game and Fresh Water Fish Commission.

_____. 1977b. The effects of grass carp introduction on water quality of four Florida Lakes. In: Florida Game and Fresh Water Fish Commission. 1977. The Grass Carp: A Special Research Report to the Governor and the Cabinet. Tallahasee, FL: Florida Game and Fresh Water Fish Commission.

Gasaway, R. D., and T. F. Drda. 1977. Effects of grass carp introduction on waterfowl habitat. Transactions of the 42nd North American Wildlife and Natural Resources Conference. 1977: 73-85.

Ghani, M. A. 1976. Insects and other phytophagous organisms which attack Florida elodea (*Hydrilla verticillata*) in Pakistan. Final Report. Commonwealth Inst. Biol. Control, Pakistan Station, Rawalpindi, Pakistan. 33 pp.

Kilgen, R. H., and R. O. Smitherman. 1971. Food habits of the white amur stocked alone and in combination with other species. Progressive Fish-Culturist 33(3):123-127.

_____. 1973. Appendix F. Food habits of the white amur (*Ctenopharyngodon idella*) stocked in ponds alone and in combination with other species. In Ganstad, E. O. (Ed.). 1973. Herbivorous Fish for Aquatic Plant Control. Aquatic Plant Control Program. Technical Report 4. Vicksburg, MS: U.S. Army Engineer Waterways Experiment Station, pp F1-F13.

Kobylinski, G. J., W. W. Miley, II, J. M. Van Dyke, and A. J. Leslie, Jr. 1980. The effects of grass carp (*Ctenopharyngodon idella* Val.) on vegetation, water quality, zooplankton, and macroinvertebrates of Deer Point Lake, Bay County, Florida. Tallahassee, FL: Bureau of Aquatic Plant Research and Control, Florida Department of Natural Resources. 114 pp.

Legner, E. F. 1979. Update of biological aquatic weed control with fish in irrigation canals in California's lower desert. In: Abstracts of 1979 Meeting of the Weed Science Society of America. 62-63.

Lembi, C. A., and B. G. Ritenour. 1977. The white amur as a biological control agent for aquatic weeds in Indiana. Technical Report No. 95. West Lafayette, In: Purdue University Water Resources Research Center. 95 pp.

Lembi, C. A., B. C. Ritenour, E. M. Iverson, and E. C. Forss. 1978. The effects of vegetation removal by grass carp on water chemistry and phytoplankton in Indiana ponds. Transactions of the American Fisheries Society 107(1):161-171.

Linchevskaya, M. D. 1966. The diet of grass carp fry. Dokl. Akad Nauk SSSR 166(3):729-731. (Weed Research Organization Annot. Bibl. No. 31: 2706.)

Michewicz, J. E., D. L. Sutton, and R. D. Blackburn. 1972. The white amur for aquatic weed control. Weed Science 20(1):106-110.

Miller, A. C., and J. L. Decell. 1984. Use of the white amur for aquatic plant management. Instruction Report A-84-1, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Miller, A. C., and R. H. King. 1984. Large-scale operations management test of use of the white amur for control of problem aquatic plants. Synthesis Report. US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Mitzner, L. 1975. Standing crop of vascular plants, primary productivity and water quality at Red Haw Lake. In: Mitzner, L. 1975. Evaluation of Biological Control of Nuisance Aquatic Vegetation by White Amur. Study No 504-1. Iowa Conservation Commission, Fisheries Section. Job No. 1. pp. 37-50.

Nall, L. E., and J. D. Schardt. 1978. Large-Scale Operations Management Tests of use of the white amur for control of problem aquatic plants: Vol. I. "The Aquatic Macrophytes of Lake Conway, Florida." Technical Report A-78-2, U.S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Miss.

National Academy of Science. 1976. Making aquatic weeds useful. Some perspectives for developing countries. Report of an Ad Hoc Panel of the Advisory Committee on Technology Innovation Board on Science and Technology for International Development Commission on International Relations. 175 pp.

Opuszynski, K. 1979. Weed control and fish production. In: Shireman, J. V. (Ed.). 1979. Proceedings of the Grass Carp Conference. Gainesville, FL: Aquatic Weeds Research Center, University of Florida, Institute of Food and Agricultural Sciences. pp 103-138.

Osborne, J. A., and M. M. Sassic. 1979. Biological control of *Hydrilla verticillata* Royle with Grass carp (*Ctenopharyngodon idella* Val.). Journal of Aquatic Plant Management 17:45-48.

Pemberton, Robert W. 1980. Exploration for natural enemies of *Hydrilla verticillata* in eastern Africa. Miscellaneous Paper A-80-1, US Army Engineer Waterways Experiment Station, Vicksburg, Miss. 23 pp.

Pennington, J. C. 1985. Biological control of *Hydrilla verticillata* (L.f.) Royle with lytic-enzyme-producing microorganisms. Technical Report In Preparation, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss. 45 pp.

Pierce, B. A. 1983. Grass carp status in the United States: A review. Environ. Manag. 7(2):151-160.

Prowse, G. A. 1969. The role of cultured pond fish in the control of eutrophication in lakes and dams. Verh. Internat. Verein. Limnol. 17(2):714-718.

Rao, V. P. 1969. Evaluation of natural enemies associated with witchweed, nutsedge and several other aquatic weeds occurring in India. U.S. PL-480 Project, Tech. Rpt. 16 pp.

Rao, V. P., and T. Sankaran. 1974. Evaluation of natural enemies associated with witchweed, nutsedge and several other aquatic weeds occurring in India. U.S. PL-480 Project. Final Tech. Rpt. 66 pp.

Rottmann, R. W. 1976. Limnological and ecological effects of grass carp in ponds. M.S. Thesis. Columbia, MO: University of Missouri. 62 pp.

Rottmann, R. W., and R. O. Anderson. 1976. Limnological and ecological effects of grass carp in ponds. Proceedings of the Southeastern Association of the Game and Fisheries Commission 30:24-39.

Rozmanova, M. D. 1966. The feeding of white amur larvae kept in fish-cages. Dokl. Akad. Nauk SSSR 166(3):729-731. (Translated from the Russian in Journal of the Biological Academy of the USSR 166(3):27-28.)

Singh, C. S. 1976. Aquatic weeds in the fisheries ponds and their control measures. In: Varshney, C. K. and J. Rzoska (Eds.). 1976 Aquatic Weeds in South East Asia. The Hague, Netherlands: Dr. W. Junk. pp 331-334.

Singh, S. B., S. C. Banerjee, and P. C. Chakrabarti. 1967. Preliminary observations on response of young Chinese carp to various physicochemical factors of water. Proceedings of the National Academy of Sciences of India 37B(3):320-324.

Singh, S. B., R. K. Dey, and P. V. G. K. Reddy. 1976. Some additional notes on the piscivorous habits of the grass carp (*Ctenopharyngodon idella*). Aquaculture 9(2):195-198.

Smith, C. R. and J. V. Shireman. 1983. White amur bibliography. Miscellaneous Paper A-83-7, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Varghese, G., and Gurchan Singh. 1976. Progress in the search for natural enemies of *Hydrilla* in Malaysia. In: Aquatic weeds in S.E. Asia. Proceedings of a regional seminar on noxious aquatic vegetation, New Delhi 1973. 341-352.

Vinogradov, V. K., and Z. K. Zolotova. 1974. The effects of the grass carp on the ecosystem of waters. Gidrobiol. Zh. 10(2):90-98. (Translated from Russian in Hydrobiol. J. 10(2):72-78.)

# CHAPTER IV: MECHANICAL/PHYSICAL CONTROL TECHNOLOGY

## Overview

For the purposes of this document, a mechanical/physical control technique is defined as any technique that directly or indirectly interferes with the growth or spread of an aquatic plant through mechanical or physical manipulation of the plant or its habitat. Under this broad definition, a wide variety of techniques could be included in this category: cutting and harvesting of aquatic plants; placement of bottom-covering material over aquatic-plant-infested sediments; dredging aquatic-plant-infested bottom sediments; mechanical agitation of bottom sediments to detach root material; confinement or collection of detached plant material; temporary reduction of water level to expose and kill root material within the sediments; and addition of light-shading material (dye) to the water. It should be apparent from an examination of the list of possible control techniques that not all techniques would be appropriate for all environments or for all aquatic-plant-control objectives.

In this chapter, only those mechanical/physical techniques that might be capable of controlling *Hydrilla* in the Potomac River are described. Those that at present appear to be operationally feasible under the operational and environmental constraints specific to the freshwater estuarine portion of the Potomac River are identified, and operational and environmental considerations associated with these techniques are discussed. Techniques that may become operationally feasible with additional development are identified, as are specific gaps in the current state-of-the-art knowledge of mechanical/physical control techniques or their interaction with monoecious *Hydrilla*. Conclusions concerning the use of presently operational techniques applicable to the Potomac River and on research necessary to fill current knowledge gaps are presented at the end of this chapter.

## Cutting And Harvesting

Cutting and harvesting of rooted submersed aquatic vegetation is the most widely used form of mechanical/physical control. All production cutters and cutter/harvester units use reciprocating sickle bars to sever plants from

their attachment to the bottom. The cutter bar system usually consists of a horizontal cutter bar 5 to 10 ft in width and two vertical cutter bars mounted on a rectangular frame that can be lowered into the water to any selected depth up to a usual maximum of around 5 ft. When a cutter unit alone is used (i.e., a machine only capable of cutting and not of harvesting or collection), then other collection techniques, either passive or active, must be used to contain and remove the cut plant material. A description of techniques for removal of detached plant material is presented later in this chapter.

All off-the-shelf cutter/harvester units capable of harvesting rooted aquatic vegetation are essentially identical in concept and sequence of operations. Each unit consists of a pontoon-mounted conveyor system propelled by paddle wheels. The leading edge of the system, which can be lowered into the water, contains the cutter bar system to detach plant material in the path of the harvester. The cut plant material immediately enters a moving conveyor system and is deposited into a holding area. Because cutting and harvesting are accomplished by the same unit, release of cut fragments to the water is minimal under favorable operational conditions (see "Conducting Control Operations" in this chapter). When the holding area is filled to capacity, the harvesting activity is suspended, and the harvester must either offload onto a waterborne transport barge or it must deliver the plant material to a takeout point for disposal. The shoreline takeout point may then serve as the final disposal site, or the material may be trucked away to an upland disposal area. Readers are referred to Newroth (1974), Canellos (1981), and Livermore and Koegel (1979) for a more comprehensive description of cutting and harvesting equipment and operations.

Harvesting tests conducted on the Potomac River during October 1984 demonstrated that harvesting is a feasible control technique in this environment and that there are no unique mechanical problems associated with harvesting monoecious *Hydrilla*.

Effects on target plants

Cutting and harvesting of rooted submersed aquatic vegetation act to immediately remove nuisance-level growth from the water column. Long-term effects on the target plants generally appear to be minimal because plants usually regrow within a period of time. Conventional harvesting should be considered only a short-term management technique (Newroth 1983). There is some evidence, however, to suggest that specific cutting applications might

achieve reduced levels of plant regrowth (Kimbel and Carpenter 1979, Barko unpublished data*). The possible value of using harvesting to achieve long-term control of *Hydrilla* is examined in this section. Also discussed are the potential effects cutting may have on species composition.

Long-term control. The physiological effects of cutting rooted submersed aquatic vegetation has only been considered for *Myriophyllum spicatum* (Kimbel and Carpenter 1979); the frequency, timing, and depth of cutting were shown to influence the amount of subsequent regrowth. Regrowth was less when cuttings were performed deep, frequently, or late in the growing season. This was attributed in part to the quantity of stored nonstructural carbohydrates remaining within the plant (Titus et al. 1975). Cut plants regrew when remaining tissues contained enough stored nonstructural carbohydrate to place a sufficient amount of photosynthetic tissue into the photic zone. Cut timing will influence regrowth since it determines the remaining time available for replenishment of carbohydrate reserves; early season cuts may allow adequate time whereas late season cuts may not. Cutting frequency affects regrowth by determining the relative amount of time the plant is expending for regrowth versus producing carbohydrates. Long-term effects of cutting may be observed if late season biomass is kept low, allowing little nonstructural carbohydrate to be shunted into storage organs prior to dieback. Plants containing low levels of nonstructural carbohydrates late in the season may not show vigorous growth the following season (Barko unpublished data*).

a. Hypothesis. Monoecious *Hydrilla* is able to overwinter primarily through tuber formation, described in Chapter II: Ecology. Tubers start forming by the last week in June and continue forming as long as live above-bottom biomass is present. The amount of photosynthetic tissue required for monoecious *Hydrilla* to form tubers is minimal; rooted stems 10 to 15 cm in length, grown under short photoperiods, could produce tubers and apparently all production goes to tuber formation, as the stems do not appear to grow. Monoecious *Hydrilla* tubers are considerably smaller than dioecious *Hydrilla*; if this also corresponds to less stored carbohydrates, then it is possible that monoecious tuber viability may not be more than one season. Additionally, it is suspected that these tubers may be the only important

---

* Dr. John W. Barko, US Army Engineer Waterways Experiment Station, Vicksburg, Miss., 1984.

overwintering structure in monoecious *Hydrilla*. If this is the case, then the plant could possibly be eliminated from an area by a cutting program that prevented tuber formation for only one or two seasons.

b. Reality. A harvesting program to effect long-term control of monoecious *Hydrilla* would necessitate cutting all plant stems to less than 10 cm (how much less is not known) by the end of June and to keep them in that condition for the remainder of the growing season. These requirements, however, are not operationally realistic using a cutter/harvester unit; such close cuttings could not be achieved initially, much less maintained. Additionally, monoecious *Hydrilla* does not exhibit a canopy-forming growth habit. Therefore, most of its above-bottom biomass is near the bottom as opposed to the top of the water column. Consequently, even a near-bottom cut might not remove a large portion of the biomass, leaving sufficient tissue for tuber production.

Shift in species competition. A final aspect of cutting effects on target vegetation that bears mentioning is possible shifts in species composition. Many sites in the Potomac contain a mix of plants including canopy-forming species, such as *Myriophyllum spicatum* and *Heteranthera dubia*, in addition to noncanopy-forming monoecious *Hydrilla*. Cutting in such areas could result in the removal of a greater portion of the biomass of canopy-forming species than of noncanopy-forming species, possibly resulting in a competitive advantage for *Hydrilla*.

Effects on nontarget organisms and on the ecosystem

Cutting and harvesting of submersed vascular plants has been described as an aquatic-plant management technique relatively free of the environmental side effects associated with other operational techniques such as dyes, dredging, sand and gravel blankets, reservoir drawdown, and use of herbicides (Carpenter and Adams 1977). Numerous researchers (Nichols 1973, Wile 1975, Peterson et al. 1974, Carpenter 1976, Pearson and Jones 1978, Breck and Kitchell 1979, Bartell and Breck 1979, Crowder and Cooper 1979, Boyle 1979, Haller et al. 1980, Swales 1982, Mickol 1984) have examined the environmental effects of cutting and harvesting on aquatic systems. Many of the effects attributed to harvesting in these studies were simply the contrast between the state of the aquatic system with and without submersed aquatic vegetation and would likely occur following the use of any aquatic-plant control technique.

Since the direct and indirect effects of aquatic vegetation on the physical, chemical, and biological characteristics of aquatic systems have been described in the chapter on ecology, presence/absence types of effects are not discussed in this chapter. Instead, discussion is focused on the direct ecosystem effects resulting from the operation of a harvester and those that are unique to cutting/harvesting among aquatic vegetation management techniques. These effects include the following: immediate physical perturbation on the system by the harvester's operation, leaching of damaged plant tissues, removal of nutrients from the aquatic system, and direct removal of nontarget fauna.

Carpenter and Gasith (1978) examined the combined water-quality effects resulting from physical perturbation of the littoral zone by a harvester's operation (resuspension of sediment, detritus, and epiphytes) and from the leaching of cut plant tissues. A detectable but short-lived increase in concentrations of suspended material, soluble organics, and soluble and particulate nutrients was observed in a harvested plot relative to an uncut reference plot. They concluded that harvester operation caused little immediate detriment to the littoral environment and that harvesting of small areas would be unlikely to significantly impact water quality.

Removal of aquatic vegetation from a water body by harvesting has been suggested as a means for removing nutrients from the water body, thereby reversing trends toward eutrophication (Livermore 1954, Hasler 1969, Livermore and Wunderlich 1969, Steward 1970). Field tests of this concept at large-scale harvesting operations have shown varied results (Peterson et al. 1974, Wile 1975). In water bodies receiving little cultural enrichment and containing much aquatic vegetation, harvesting a large portion of the vegetation can remove a significant portion of the water body's annual nutrient loading (Wile 1975). In contrast, for aquatic-plant-infested lakes receiving cultural enrichment, harvesting is not likely to appreciably reduce net annual nutrient loading (Peterson et al. 1974, Burton et al. 1979). Within the Potomac River, harvesting of aquatic vegetation would probably not result in any significant reduction in net nutrient loading.

Many types of aquatic fauna live in close association with aquatic vegetation (see chapter on ecology). Cutting and harvesting will unavoidably remove a certain portion of these organisms. Pearson and Jones (1978) examined the effects of cutting and subsequent passive collection of aquatic

vegetation within an English Chalk stream on macroinvertebrate fauna. They observed that cutting greatly increased the drifting of these organisms. Redistribution patterns were different between species, and no generalized response appeared to occur. Faunal recolonization of the regrowing plants was rapid. No quantitative estimates were made of the portions of each invertebrate population removed by harvesting. They concluded that the long-term effects of cutting on the macroinvertebrate community were minimal, although it was speculated that timing of the cuts in relation to the emergence of dominant aquatic insects could have a significant effect on organisms using these insects as a food source.

Several researchers have studied the direct effects of harvesting on fisheries (Wile 1978, Haller et al. 1980, Mikol 1984). Wile (1978) estimated that direct loss of fish entrapped in milfoil harvested from an Ontario lake was approximately 1 part fish to 1000 parts plant on a wet-weight basis, or 8.9 kg fish/ha harvested. No attempt was made to quantify the portion of the fisheries' standing crop removed by harvesting. Over the study period, fish populations either remained stable or changed in ways that were not attributable to harvesting.

More detailed examination of the direct effects of harvesting on fisheries was made by Haller et al. (1980) during harvesting of dense *Hydrilla* in Florida and by Mikol (1984) for harvesting relatively sparse milfoil in New York. In each study fish densities and biomass by species removed by harvesting were compared with those for otherwise similar unharvested areas. The estimated percentage of fish (number) was found to be 2.5 percent by Mikol, and 32.0 percent by Haller. There are undoubtedly a number of factors contributing to this wide variation. Both studies indicate that the percent capture varies greatly between fish species and different sizes; smaller fish are most heavily impacted as are certain species which apparently exhibit the wrong responses to avoid a harvester. It seems likely that plant species and density are also important factors: the denser the vegetation, the less would be the chances of escape. The 32.0-percent capture rate observed by Haller occurred in dense (11 tons/acre) *Hydrilla*, whereas the 2.5-percent capture rate observed by Mikol occurred in sparse (3 tons/acre) milfoil.

For all direct effects cited by Heller and Mikol and for all effects associated with the presence/absence of aquatic vegetation, the most important factor in assessing the level of impacts on an aquatic ecosystem is the

magnitude of the harvesting operation. Even seemingly minor effects may be detrimental to the aquatic ecosystem if all or most of the vegetation is harvested. For this reason, it is suggested that the area treated by cutting and harvesting be kept to the minimum amount required to satisfy the needs of water-body users. Techniques for doing this are discussed in the next section.

Planning control operations

An off-the-shelf harvesting system usually consists of a cutter/harvester unit and a shoreline conveyor for offloading; in some cases, additional barges to transport the material to shore can be included. The purchase price for off-the-shelf cutter/harvester units is in the range of approximately $20,000-$85,000; specialized shore conveyors and separate transport units may add another $50,000 to the price of the total system. For contracting, hourly rates for equipment with operators may range from $70 to $250 per hour, depending upon the specific equipment used and the location of the operation.

Given the high cost of purchasing or renting a harvesting system, it is imperative that a sound operational plan be developed and implemented in order to ensure effective use of aquatic plant control funds. A number of harvesting operations targeted toward *Hydrilla* have been successful in maintaining the plant below nuisance levels; other programs have proved to be less successful. The difference between success or failure of an operation is largely determined by the amount of effort that goes into developing and implementing a plan. The steps involved in developing a sound plan include:

1. Defining specific areas to be harvested and harvesting pattern.

2. Determining the required schedule.

3. Identifying shoreline takeout points and upland disposal sites (if necessary).

4. Determining state or local permit or statutory requirements for harvesting and disposal operations.

5. Defining the specific mix of equipment most likely to be cost-effective for the situation.

6. Obtaining or contracting for the equipment.

Step 1. A high density of aquatic plants within an area does not automatically constitute a nuisance situation. Nuisances are created only when plants in a specific area interfere with a specific water-body use in that

area. The principal aquatic-plant-related nuisance situation on the Potomac River is interference with recreational boating; areas requiring treatment will be those with high plant densities and with high recreational boating activity levels. Once areas needing treatment have been identified, specific harvesting plots or patterns should be defined within these areas.

Harvesting can be used for vegetation removal in large areas; however, this is generally not cost effective. Instead, planners should strive to define a harvesting pattern that maximizes relief from nuisance growth while minimizing the area harvested. For recreational boating, the ideal harvesting pattern is usually a series of access channels that provide boaters access to and from deep open-water areas. If an area is known to be frequently subjected to winds or currents from a particular direction, an attempt should be made to align harvesting patterns in this direction since operating at an angle to it may degrade harvester performance (discussed in the next section). Also, it is advisable to lay out several alternative patterns involving different amounts of harvesting area; this way, at least one pattern would be acceptable given different funding levels. Using access channel patterns for harvesting allows a larger number of sites to be treated with a fixed amount of money than would be possible for harvesting large contiguous areas.

Step 2. The harvesting schedule (i.e., when and how often to harvest) may be determined by combining known information on the growth cycle of the nuisance plants with the timing of the demand for the impeded water-body use. Since quantitative information on the growth cycle of monoecious *Hydrilla* is presently limited, it will be necessary to rely on observations of boaters and marina operators until quantitative information and predictive capabilities become available. The timing of water-body use by recreational boaters will expectably have a cyclic pattern with certain distinct peaks. For instance, usage levels will be much higher on the weekends than during the week, and certain weekends (Memorial Day, July 4, and Labor Day) will have much higher usage than other weekends. Armed with this information, the planner can establish a schedule that will provide relief from nuisance aquatic plant growth at the high usage areas during peak usage times.

Step 3. The next step is to locate all prospective shoreline takeout points and upland disposal sites in the vicinity of a harvesting area. Actual requirements for a shoreline takeout point will vary depending on the specific vessel used for transporting harvested plant material. As a general rule,

however, if water at the shoreline is at least 2 ft deep within a 5 ft distance from the shoreline, then even the larger harvesters would be able to reach the shore. Additionally, there should be road access to the shore takeout point such that a mobile conveyor could be positioned at the shoreline and large dump trucks would have access. Once takeout points are identified, efforts must be made to secure the cooperation of property owners at all potential sites since this will add to the flexibility and efficiency of the harvest. It will also provide backup sites if one property owner decides against use after the project begins.

The final disposal site for harvested material could be at shoreline takeout points in underdeveloped areas. Operationally this is highly desirable because it avoids an expensive and frequently delay-causing trucking operation. This is not often the case, however, since most harvesting sites will be in high-use recreational areas where adjoining shoreline will be developed. In moderately developed areas, it may be possible to find a nearby farmer, commercial nursery owner, or landowner willing to allow plant material to be deposited on their property. Since submersed aquatic plant material is 80- to 90-percent water and contains very little lignin, it rapidly desiccates and decomposes to a relatively minuscule volume within several warm-weather months. In highly developed areas (such as the Alexandria, Va., waterfront), it may be impossible to locate a nearby upland disposal site; and even if a site could be located, it would require transport on heavily traveled roads through populated areas, which would be undesirable. In such a case, the only suitable solution may be to use a commercial refuse hauler. A dumpster could be placed under the shoreline conveyor and harvested material deposited in it. A 30-cu-yd dumpster would hold approximately eight loads from a medium-sized harvester, which should be sufficient capacity for a full day's operation. While this is not inexpensive (adding approximately $300/day to the operational cost), it may be the only feasible option in some situations.

In order to identify alternatives to disposing of harvested plant material, extensive research has been conducted to find productive uses for the material. Research studies on productive uses of Florida dioecious *Hydrilla* have examined composting, production of cattle feed, and production of methane gas. Composting has been shown to reduce *Hydrilla* to a minuscule volume and to produce good quality compost material within several months. Tests of cattle feed produced from harvested *Hydrilla* have shown that feed

quality and acceptability vary greatly with the source of the *Hydrilla*. Methane production rates from anaerobic digestion of *Hydrilla* have been shown to be sporadic relative to waterhyacinth. These studies would indicate that the prospects for identifying an economic use of *Hydrilla* are poor. Even if such a use were identified, it would be more economical to culture the *Hydrilla* at a production facility in order to produce a uniform quality raw material and to avoid the high cost and logistical problems associated with transport of *Hydrilla* from a natural site. Disposal operations, therefore, appear to be an unavoidable step in planning a harvesting operation.

Step 4. A consideration at this point in the planning process should be the possible necessity of obtaining permits required by state or local governmental agencies to conduct any phase of the overall operation. No state in the US currently has any permit or statutory requirements regulating aquatic-plant harvesting (with the exception of Wisconsin, which requires that all cut aquatic plants be removed from the water); however, this may change in the near future as there is legislation proposed in Maryland to regulate aquatic-plant clearing.

Step 5. The overall effectiveness and the cost-effectiveness of harvesting programs are most dependent on obtaining the right type and mixture of equipment for the particular harvesting site. The next step is to determine this type and mixture. The overall productivity of a harvesting system (tons harvested/hour or acres harvested/hour) is determined by the simultaneous interaction of all the processes involved in the operation. For any specific harvesting operation (i.e., a specific set of environmental and operational conditions and mechanical performance parameters), the system productivity will be limited by the slowest step in the process, which will expectably vary among sites and types and mixes of equipment.

Rule-of-thumb estimates do not take into account important interactions between plant density, takeout point location, and mechanical performance parameters and may result in large errors in cost and productivity estimation (see Smith (1979) for examples and discussion of the variability of operational costs). Manual means of simultaneously keeping track of all these interactive variables would be too cumbersome; therefore, the computer model HARVEST (Sabol and Hutto 1984) was developed at WES for this purpose.

HARVEST is a simple deterministic model that can simulate mechanical control operations in submersed or attached floating aquatic plants in any

realistic operational environment. It can be used to simulate some or all of the following specific mechanical operations: cutting, collection, onboard processing, transport to a shore takeout point, and trucking to a final upland disposal site. Each important step in an operation is simulated, and estimates of time and cost are computed.

The HARVEST model and associated documentation have been transferred to the US Army Engineer District, Baltimore, and will not be discussed in detail in this document. In the planning phase, HARVEST can be used to estimate costs and to select the type and mix of equipment to be used by comparing the cost and productivity of alternate mechanical systems operating at the different selected sites. Additionally, it may be used to determine how changes in a machine or operational conditions affect productivity rates; this mode of use can identify means of making an operation more efficient. A detailed example of the use of HARVEST for planning a harvesting operation is described by Sabol (1983).

Step 6. The final phase of the planning procedure is to obtain the equipment for an in-house effort or to contract for harvesting services. An in-house operation requires that personnel be trained and assigned to operate the equipment and that mobile repair facilities be set up for onsite maintenance and repair of the equipment. In temperate climates, where aquatic plants die back during the winter (such as the Washington, D.C., area), the equipment-operator jobs will only be seasonal. Personnel to fill these positions must either be hired seasonally or assigned to alternate duties during the winter. This makes it difficult to obtain and keep the experienced operators who are essential for efficient use of harvesters. Additionally, when harvesting is performed by in-house personnel, it is difficult to respond quickly to variable levels of harvesting needs. For a governmental agency, therefore, contracting may be preferable to an in-house effort since it alleviates problems with keeping experienced operators and it offers flexibility in planning and conducting mechanical control operations.

If the operations are to be contracted, several factors need to be considered in preparation of the bid specifications. First, the best type and mix of equipment should be determined using the HARVEST model. This particular equipment should then be required in the specifications by referring either to the make and model of equipment or to minimum acceptable characteristics such as cutter width and storage capacity. Additionally, it is

imperative that the individual pieces of equipment work together as a system. This can usually be handled in the specifications by stating make and model of equipment. Any nonstandard equipment proposed for use by a contractor should be carefully examined before a bid is accepted to ensure its compatibility with the overall system. If the project is for the duration of a growing season, a 40-hour work week may be used. As setup time each day includes moving from the night parking site to the harvesting area and back, there is some unproductive time each day. One method that increases the amount of productive time on the job is to work four 10-hr days per week. This has been found to be operationally effective. This also gives the contractor 3 days per week when he can perform work for private groups whose property borders the management area. This extra revenue may allow the contractor to offer his services to the district at a lower rate. In a contract for a long-term harvesting operation, there should be performance goals the contractor is required to meet and monetary penalty for nonperformance. This will force the bidder to fully consider the operation and his ability to perform for the price he quotes. It should be further required that the bidder review the site with a representative of the district to get a clear picture of what he will be required to do.

Conducting control operations

Several considerations need to be taken into account during a harvesting operation in order to ensure its success. These considerations include: strategies for minimizing downtime; methods to compensate for effects of environmental and operational conditions on system performance and mode of operation; and techniques for minimizing release and for disposal of plant fragments.

Downtime. Any harvesting operation, however well planned, will not succeed if excessive amounts of downtime occur. For an in-house operation, downtime will result in the continued expenditures of funds without accomplishing any aquatic-plant control. For a contract operation, the financial effects of downtime to the district will not be severe if the contract specifies only partial or no payment for downtime, as it should. However, downtime still represents lost time that may translate into a reduced area of control. Downtime can be minimized through careful attention to maintenance procedures, by developing means of making rapid repairs when equipment failure or damage does occur, and through careful use of the equipment by its operators. A contract

for harvesting operations should require such actions, and the operation should be monitored for compliance.

A daily maintenance routine will alleviate much of the need for field repairs. Before operations begin each day, all grease fittings should be lubed, especially those at or below the water's surface. The belting should be inspected and repaired as necessary. Proper safety equipment should be available on the site. It is important to have a secure area to park the units at night in order to prevent vandalism.

The operation should have onsite repair capability to minimize downtime. Spare parts on hand at the site should include: belts; filters for the hydraulic, gas, and oil systems; hydraulic oil; harvester knives and rock guards; pitman rods and ends for the drive system of the knives; and slide guards for the knife assembly. At the beginning of the season, the operator should develop a list of sources for onsite hydraulic repair and welding. Arrangements should be made with the equipment manufacturer to have express freight available for other parts such as hydraulic components. If a contractor is involved, the above should be specified within the contract.

Environmental and operational conditions. These conditions can greatly affect the performance of a harvester. Harvester performance is best in the absence of wind and current; increasing amounts of one or both factors will result in a progressive degradation of equipment performance until a point is reached at which the harvester is no longer effective. Unfortunately, relatively few harvesting days or sites will have optimal conditions for harvesting; therefore, it is usually necessary to operate on any day or in any place that performance will be above a minimum acceptable level.

Paddle-wheel-propelled harvesters are awkward crafts to maneuver and have large cross-sectional areas above the waterline that are subjected to wind forces. Winds or water currents acting at an angle to the harvester's working path will force the operator to continually take corrective action (reversing and running each paddle wheel at a variable speed) to maintain a straight path. This creates irregular motion of the harvester and additional turbulence that act to interrupt the smooth flow of plant material onto and up the conveyor; thus harvester performance (plant material collected/unit time) will decrease. Depending upon the specific harvester used, wind speed above 10 to 15 mph or water current speed above 0.25 mph can degrade harvester performance when acting perpendicular to the harvester's path. If performance

becomes severely degraded, the only option other than stopping the operation until conditions improve is to align the harvesting path with the direction of wind and/or water current. This may or may not be possible depending upon the required harvesting pattern; however, if it is known before hand that strong water currents or prevailing winds occur in a particular location, then it may be possible to select a harvesting path that meets the needs for vegetation control and is aligned with the direction of winds or water flow.

Submersed aquatic vegetation acts like a wind sock in response to water currents. This is particularly apparent in the Potomac River during ebb and flood tides. The quality of the cut achieved by a harvester is sensitive to the directional orientation of submersed aquatic vegetation. In strong current situations, harvesting in the direction of the flow will result in failure to remove a large portion of the vegetation within the harvesting depth. On the contrary, harvesting against the direction of flow will result in very effective removal of vegetation within the harvesting zone; this practice should be followed when strong currents are encountered.

Tides within the Potomac River, in addition to creating currents, cause a 2- to 3-ft water-level fluctuation that must be accounted for in the daily operating plan. Most harvesting units cut only to a depth of 5 ft. To use the water-level fluctuations to maximize effective use of the equipment, shallow water areas should be cut during high tide and the deeper water areas during low tide. Tidal fluctuation will also affect the shoreline takeout operation. The shoreline conveyor will need to be continually repositioned throughout the tidal cycle, and, in some cases, certain takeout points may not be visible during low tide.

Some of the effects that environmental and operational conditions are likely to have on harvester performance can be taken into account during the planning stage if conditions at a site have been adequately described. Other conditions, such as inaccessibility of a shoreline takeout point at unusually extreme low tides or wind conditions on a particular day, may be overlooked or not foreseeable. It is possible to have developed a specific plan for harvesting that is incompatible with environmental or operational conditions at that site or at a particular time. Adherence to such a plan would result in ineffective harvester operation. It is necessary for an in-house harvester operator or an onsite contract monitor to have the flexibility to vary the operational plan if needed to achieve effective use of the equipment. So long

as the responsible individual is aware of the purpose and objectives for harvesting a particular site, necessary onsite modifications should produce acceptable results.

Plant fragments. Left in a water body, fragments of *Hydrilla* will become rooted and produce new plants. Under ideal harvesting conditions (i.e., no wind or water flow) or when the harvesting path aligns with direction of the wind or water currents, the flow of plant material onto and up the conveyor system can be a smooth uninterrupted process that results in negligible release of plant fragments. Most harvesting operations, however, do result in some fragmentation. It is necessary to use various techniques to prevent dispersal of floating fragments. The least time-consuming techniques are those that take advantage of the conditions at the site and require no extra equipment or effort. One such technique is to use surface-topped aquatic vegetation as a barrier or filter to contain most of the fragments. To employ this technique, harvesting operations are conducted in the middle of a large contiguous area of surface-topped vegetation; the perimeter vegetation acts to temporarily contain fragments generated by harvesting. After the central portion of the area has been harvested and winds or water currents have driven fragments into the perimeter vegetation, this vegetation is harvested, which removes most of the collected fragments. If it is not possible to use this technique, it may be possible to harvest upstream or upwind of an area of surface-topped vegetation to achieve the same effect. Timing will be critical in this technique since flows cyclically reverse and winds may be transient. Both of these techniques require good onsite judgment by the harvester operator.

As a last resort, a floating boom, that is described in a later section, "Confinement and Collection of Detached Plant Fragments," may be strategically placed to collect floating fragments. After harvesting is completed and floating fragments have had time to collect against the boom, the fragments can be manually removed from the boom. This technique, in addition to requiring a boom, requires extra time to place, clean, and retrieve the boom. Good onsite judgment is also required in determining the required location of the boom.

## Bottom-Covering Materials

Bottom coverings refer to a broad class of materials that can control the growth of rooted aquatic plants by physical alteration of the plant's environment such that existing plants may die and decay and/or new growth may be inhibited. Bottom-covering materials that have been tested for control of aquatic vegetation include sand and gravel, sand and gravel laid on impermeable membranes, impermeable membranes, and various types of porous fabric materials. The use of sand and gravel has been shown to provide only short-term control and is judged to be ineffective considering its cost and installation problems (Engel and Nichols 1984). Permeable bottom-covering fabrics and impermeable bottom-covering membranes have been shown to be generally more effective (Armour et al. 1979, Cooke 1980) and are discussed in this section. A fabric material and a membrane material were installed at test sites within the Potomac River during 1984; present indications are that this may be a feasible technique for controlling *Hydrilla* within the Potomac River.

### Factors affecting extent and duration of control

Bottom-covering fabrics and membranes can control the growth of rooted aquatic vegetation by a number of mechanisms that can act alone or in combination, depending upon the environmental conditions and the material applied. These mechanisms include:

1. Reducing light levels to the point where plants cannot meet their photosynthetic requirements (Bulthuis 1984).

2. Compressing and confining existing plant tissue into a reducing chemical environment that stresses and kills the plants and leads to their subsequent decay by enhanced microbial activity at the sediment interface (Perkins, personal communication*).

3. Providing a physical barrier through which plants cannot penetrate and which is unsuitable as a substrate for recolonization of rooted vegetation (Boston and Perkins 1982).

In addition to the material affecting the environment, chemical and physical processes will affect the emplaced material and determine its immediate and long-term effectiveness. These environmental-based processes include:

---

* Michael A. Perkins, University of Washington, Seattle, Wash., 1985.

1. Degradation of materials exposed to ultraviolet light.

2. Decomposition of material by microbial activity.

3. Deposits of sediment on top of the material leading to recolonization by rooted vegetation.

4. Gas production within the sediment that can build up under impermeable membrane (or permeable membrane that becomes impermeable through sedimentation), causing the membrane to lose contact with the bottom (ballooning) and become ineffective.

5. Current and water action that can dislodge the emplaced material from the bottom.

Types of material available

Numerous bottom-covering fabrics and membranes have been tested for control of rooted aquatic plants; a list of these is presented in Table 1. Of these materials, only Aquascreen and Dartek have received sufficiently widespread use to evaluate their performance under a range of environmental conditions. These two materials are discussed in this section.

Aquascreen. Aquascreen is a woven fiberglass screening material (62 apertures/sq cm) that allows passage of gas and light (60-percent attenuation, Mayer 1978) but prevents penetration by most types of rooted aquatic vegetation (Perkins et al. 1980). This material is heavy (specific gravity 2.54, Perkins et al. 1980) and sturdy, and it is relatively easy to install (Perkins 1984). When placed over a dense stand of submersed vegetation, Aquascreen may or may not cause the plant canopy to collapse into the bottom (Perkins et al. 1980, Pullman and Craig 1981), apparently depending on the physical characteristics of the canopy. Given that Potomac River *Hydrilla* does not exhibit a canopy-forming growth habit and does not have robust stems, it seems likely that Aquascreen would be able to force the plant into contact with the bottom.

The principal mechanism for control of established vegetation by Aquascreen is the stress placed upon the plants by confinement within a chemically reducing environment and subsequent decomposition due to high microbial activity at the sediment surface (Perkins, personal communication*). Cases in which Aquascreen has not been effective (Pullman and Craig 1981) have been attributed to failure to achieve good bottom contact. Light attenuation

---

* Michael A. Perkins, University of Washington, Seattle, Wash., 1985.

Table 1

Bottom-Covering Fabrics and Membranes Used for Aquatic Plant Control*

| Material | Description | Gas Permeable | Light Transmittance | Causes of Degradation | Specific Gravity | Duration of Control** | Testing Citation† | Cost per sq ft †† | Source |
|---|---|---|---|---|---|---|---|---|---|
| Aquascreen | Woven fiberglass screen | Yes | 60% | - | 2.5 | Not. Aua.1 | 1, 2, 3, 6, 9 | $0.30 | Menardi Southern Augusta, Ga. |
| Dartek | Black-pigmented nylon film | No (slits cut at factory) | 0% | Ultraviolet light | - | " | 4, 7 | 0.10 | Dupont Canada |
| Typar | Spun black polypropylene fiber | Yes | 0% | - | 0.95 | 1 yr | 6, 8 | 0.075 | Dupont |
| Bidim | Polypropylene/terephthalate fiber | Yes | - | - | 1.3 | 1 yr | 6 | 0.75 | Monsanto |
| Hypalon | Synthetic rubber membrane | No | 0% | - | >1.0 | - | 11 | 0.57 | - |
| Texel | White woven fiberglass | Yes | Yes | - | - | - | 5 | 0.07 | - |
| Terratrack | Woven polypropylene fiber | Yes | - | - | 0.91 | >1 yr | 3 | 0.055 | Terrafix Ltd. Rexdale, Ontario |
| Burlap | Natural fiber | Yes | Some | Microbial decay | - | <1 yr | 10 | 0.05 | - |

* All information presented was obtained from the testing citations and not from the manufacturer.

** Refers to control exerted on target plants and not economic life of material.

† Information was taken from the following sources:
1) Mayer 1978
2) Perkins et al. 1980
3) Lewis et al. 1983
4) Perkins 1984
5) Terry McNabb, Aquatic Unlimited personal communication, 1985
6) Engel 1984
7) Russell I. James, Ecoscience In. personal communication, 1985
8) Cooke and Gorman 1980
9) Pullman and Craig 1981
10) Jones and Cooke 1983
11) Armour et al. 1979

†† Includes only the cost of the material (obtained from citations) and not cost of installation.

produced by Aquascreen is insufficient to retard photosynthesis of plants with low light requirements (Mayer 1978, Perkins et al. 1980, Pullman and Craig 1981).

Aquascreen is not readily subject to microbial, chemical, or ultraviolet light degradation, and it can maintain its integrity for six or more years if not stressed physically (Perkins, personal communication*). The primary limitation on the duration of control achievable with Aquascreen is sediment accumulation; in a sediment-accreting environment, control effected by Aquascreen may only be seasonal (Engel 1984). To alleviate the effects of sedimentation, Aquascreen could be removed, cleaned, and reinstalled (Engel 1984), although the physical stress associated with this operation may reduce its effective life.

A 14- by 50-ft panel of Aquascreen was installed over standing *Hydrilla* at the mouth of Swan Creek in the Potomac River in October 1984. Monitoring of *Hydrilla* growth during the summer of 1985 will help determine the effects of Aquascreen on monoecious *Hydrilla* in the Potomac River.

Dartek. Dartek is an impermeable black-pigmented nylon film available in a 2-mil (0.05-mm) thickness. Dartek acts to control existing growth by eliminating light. Dartek may also function in a preventative mode: its surface is smooth and glossy, thus rooting aquatic plants may be unable to attach to it as has been observed with rougher fabric materials (Perkins 1984). Since the material is so thin, it behaves as though naturally buoyant in water and will not cause a standing canopy of submersed vegetation to collapse without very meticulous anchoring (Perkins 1984; James, personal communication*). To avoid ballooning caused by accumulation of gases beneath the material, the manufacturer has cut a pattern of cross-hatch slits in the material to allow gas to escape; the cuts could permit plants to grow up through the openings (Perkins 1984).

Dartek is not susceptible to chemical or microbial degradation, although it is susceptible to degradation from ultraviolet light when placed in waters shallower than 2 ft (Perkins, personal communication*). Sedimentation may limit the duration of control achievable with Dartek, although its smooth surface may allow sediments to be scoured from it more easily than from comparable rough-surfaced fabric materials. The 2-mil-thick Dartek would likely be

---

* Michael A. Perkins, University of Washington, Seattle, Wash., 1985.

destroyed by attempts to remove, clean, and reinstall it; however, this may be possible with 4-mil-thick Dartek that will soon be available (McNabb personal communication*).

Approximately 40,000 sq ft of Dartek was installed in Dike Marsh by the National Park Service in the spring of 1984. This Dartek exhibited excellent *Hydrilla* control during the 1984 growing season. Additionally, 1,600 sq ft of Dartek was placed in the Swan Creek area during fall of 1984 by Baltimore District. Continued observation at these sites will provide information on the longevity of control in this environment.

Operational considerations

Installation. The degree of vegetation control achievable with bottom-covering fabrics and membranes depends largely on obtaining and maintaining good contact between the material and the substrate. Depending upon the characteristics of the material and those of the target environment, this could be a difficult and labor-intensive task.

Most materials are available as rolls or sheets of at least 8 ft in width and of variable lengths. Panels of material may be applied as received from the manufacturer, may be joined together before application, or may be placed adjacent to each other on the bottom. To install, one end of the panel must first be secured to the bottom with pins or anchors (either bars or blocks); the panel is then unrolled or unfolded from the bow of a boat that slowly backs up to the desired end point of the panel. At frequent intervals, additional pins or anchors must be placed along the edges. This operation generally requires a small boat with operator and deck hand, and two or three divers (SCUBA equipment may not be necessary in shallow water).

Installation problems are common when applying the material over soft unconsolidated sediments or on top of a developed stand of submersed aquatic vegetation. Soft substrates can cause poor contact with the bottom as well as difficulty in securing the edges with pins. Anchors, either rods or blocks, make the securing procedure easier, but can reduce the in-situ panel width and length by crumpling of the material into the soft bottom. The anchors can also cause depressions that will catch sediments, and shorten the duration of control. Best overall installation in soft substrates can be achieved by pinning the edges, although extra efforts will be required to

---

* Terry McNabb, Aquatics Unlimited, Concord, Ga., 1985.

ensure that the pins are securely held by the substrate and that the pins secure the material without tearing.

Existing stands of submersed aquatic vegetation present similar problems. Heavy fabric materials, such as Aquascreen, can generally force the canopy to sink within several days. Light-weight material that blocks light will not sink until the plants have died, which may take several weeks. This problem can be avoided if panels are placed either immediately following harvesting, or prior to the emergence of new growth in the spring. Placement prior to spring growth requires some extra planning, since it will be necessary to locate the sites for placement of the materials based upon ground survey or aerial photo data obtained during the previous growing season.

Currents or strong tides may also pose an installation problem, although James (personal communication*) indicated that the tidal currents within Dike Marsh appeared to iron out air pockets in applied Dartek, resulting in better contact with the substratum.

The dimension of the installed material will depend upon the intended use of the water area. If materials are to be positioned to allow boat access to an area, the material should be sufficiently wide to account for the desired width of the boat lane, the amount of encroachment by the plant canopy at low tide, and the possible reduced width of the in-situ panels caused by crumpling of the material into soft sediments. Canopy encroachment can be estimated based on water depth, plant-stem length in the target area, and field examination of emplaced material.

Maintaining vegetation control. Problems associated with maintaining vegetation control after installation include degradation of the material, ballooning from entrapped sediment gases, mechanical dislodgement by the panel, and sedimentation. Materials that are easily degraded by ultraviolet light (Table 1) or that are rapidly decomposed by microbial activity should generally be avoided unless only temporary control is desired.

Production of sediment gas tends to increase with increasing organic content of the substrate and may cause ballooning problems for impermeable membranes or permeable fabrics that become clogged with sediment. The best precaution against ballooning is to ensure that panels are installed securely; upward lift caused by temporarily trapped gas will be opposed by the downward

---

* Russell I. James, Ecoscience, Inc., Old Forge, Pa., 1985.

anchoring forces, maintaining the panel's position while the gas seeps through the fabric or locates a gas vent. Mechanical dislodgement is also best prevented by careful attention to installation.

Little can be done to counteract sedimentation. Materials with smooth surfaces may experience less sediment accumulation than comparable rough-surfaced fabric materials since sediments might be scoured more easily from a smooth surface. Once a panel becomes sediment covered, its effectiveness ends unless sediments can be removed. The only bottom-sheeting material that has successfully been retrieved, cleaned, and reinstalled is Aquascreen. Other techniques that may be useful, although not reported, include sediment scouring from directionally applied prop wash and sediment removal using a diver-operated dredge (discussed later in section on dredging).

Effects on water-sediment chemistry

Sediment/water interface. Sediments are a major site of nutrient cycling and retention in aquatic ecosystems. Thus, the effect of bottom covers on the chemical and microbially mediated reactions at the sediment/water interface must be carefully evaluated with regard to water quality and productivity of the aquatic ecosystem.

Biogeochemical cycling in an aquatic system is affected by the bacterial metabolic processes functioning in the underlying sediments. Depending on the redox status of the sediment, two general types of microbial metabolism may be found: (1) processes utilizing inorganic substances (carbon dioxide, nitrogen, oxides, manganic compounds, ferric compounds, and sulfate) and (2) fermentation processes in which organic molecules are utilized as electron acceptors. Aerobic, facultative anaerobic, and obligate anaerobic metabolism will occur simultaneously in a sediment due to zonation.

Aerobic mineralization occurs at the sediment surface through the activity of the benthic microorganisms and animals that constitute an aerobic detritus food chain. The oxic zone generally constitutes a very thin layer in sediments. Complete mineralization of detritus in this zone results in the formation of oxidized inorganic species such as $NO_3$, $SO_4$, $PO_4$, and $CO_2$. Since the oxygen is quickly depleted with increasing depth of sediment, microorganisms must shift to other oxidants (electron acceptors). They do this in a sequence determined by the energy yielded by the reaction. The order of oxidants used is $NO_3$ and $Mn^{+4}$, $Fe^{+3}$, $SO_4$, and $HCO_3$. Thus, a vertical sequence of denitrifying, sulfate-reducing, and methane-producing zones can be found

in most sediments. The effect of the bottom covers on the sediment chemistry will be due primarily to changes in the oxygen status of the surficial sediment layer. When a cover is placed over the sediment, oxygen transport is either stopped completely if the covering material is impermeable to gases or is reduced to various degrees depending upon the permeability of the cover and the oxygen demand of the sediment.

The occurrence of both aerobic and anaerobic zones in the sediment is particularly important to the cycling of nitrogen and sulfur in aquatic systems. For example, nitrogen can be oxidized to $NO_3$ in the aerobic zone and then diffuse back into the anaerobic zone where it can be denitrified and lost to the atmosphere. Denitrification can be a major sink for nitrogen in aquatic systems. For example, in Narragansett Bay, R.I., about 50 percent of the inorganic nitrogen coming into the bay from land runoff and sewage was denitrified at the sediment/water interface. In addition, about 35 percent of the organic nitrogen mineralized in the sediments was removed from the bay by denitrification (Seitzinger et al. 1984). Bottom covers would likely eliminate this cleansing mechanism operating in aquatic systems by preventing the interaction between sediment and overlying water.

On the positive side, bottom covers would also eliminate or reduce the effects of the sediments as a source of nutrients to the overlying water. However, if the barrier is removed, a substantial quantity of accumulated soluble materials may be released to the overlying water. If the sediment under the cover becomes completely anaerobic, the pH is likely to decrease due to accumulation of soluble organic acids produced during anaerobic microbial metabolism. Soluble phosphorus concentrations will increase due to increased solubility of reduced iron. Lowered pH and increased concentrations of soluble organic acids could also result in greater solubility of toxic metals which had accumulated in the sediment. Ammonium concentrations in the interstitial water would also increase under the cover (Graetz et al. 1973, Byrnes et al. 1972). When the cover is removed these soluble materials (P, $NH_4$, toxic metals) could significantly affect the quality of the overlying water.

Various types of bottom-covering materials have been evaluated under field conditions (Table 1). In most cases, only effects on plant communities have been evaluated, although effects on sediment/water interchange have been suggested (Armour et al. 1979, Perkins et al. 1980, Zisette 1983). In one

case, a statement was made that "no changes in bottom sediments were observed" but this appeared to be based on a visual observation and did not address sediment chemistry. Two investigations indicated an increase in algal growth in the water column after installation of bottom covers (Cooke and Gorman 1980, Boston and Perkins 1982). This was apparently due to an increase in available nutrients in the overlying water caused by lack of macrophyte uptake or removal of the sediment as a nutrient sink.

Sediment. No research was found that directly addressed the relationship of sediment material covers to sediment chemistry. However, two investigations provided insight into possible effects of sediment covering. Engel (1984) measured the population of benthic macroinvertebrates under covers and found that their numbers were reduced two-thirds. Since the invertebrates potentially have an effect on nutrient transfer within the sediment, nutrient cycling would be affected. Boston and Perkins (1982) investigated nutrient regeneration from decaying macrophytes beneath a bottom cover and found significant effects on oxygen demand at the sediment/water interface. When macrophytes were forced directly into the sediment by the cover, death was rapid and oxygen demand was high. However, they found in the field that macrophytes were usually not forced directly into the sediment. This resulted in reduced rates of death and decomposition, with little effect on oxygen demand. They suggest applying the cover prior to plant growth in the spring to solve this potential problem. Harvesting or physical removal of vegetation would also alleviate this problem while chemical control, depending on the procedure selected, could compound the problem.

The potential effects of bottom covers on sediment biogeochemistry will depend on several factors, including characteristics of the cover (permeable or impermeable), sediment characteristics (organic matter content, toxic metal levels, nutrient levels), and the presence or absence of decaying plant material beneath the cover. The effect on the overall aquatic system will depend on its hydrologic characteristics and the relative area of sediment covered.

Effects on macroinvertebrates

Macroinvertebrate fauna are known to be generally abundant in areas containing aquatic macrophytes, both on the plants and within the sediment (see discussion on macroinvertebrates in chapter on ecology). Yet, of the numerous studies on the use of bottom covers for aquatic plant control, only one directly examined their effects on benthic macroinvertebrates. Engel (1984)

Table 2

Densities of Benthic Macroinvertebrates Under Aquascreen and in a Nearby Control Area*

| Dates | Densities in Control Area (No Cover) | Densities Under Bottom Cover | |
|---|---|---|---|
| | | Set on 5/9/80 | Set in May, 1979 |
| May 9, 1980 (before '80 cover set) | 21,000** | 21,000 | not sampled |
| May 9, 1980: | '80 COVER SET | | |
| June 13, 1980 | 14,000 | 8,000 | 1,100 |
| July 24, 1980 | 16,000 | 4,000 | 4,100 |
| Aug. 22, 1980 | 7,000 | 2,000 | 750 |

* Data from Cox Hollow Lake, Wis. (Engel 1984).
** Number of benthic macroinvertebrates per square meter of bottom surface.

found that the numbers of benthic invertebrates under Aquascreen panels in a Wisconsin Lake were reduced in comparison to nearby controls (Table 2). Although this reduction in infaunal invertebrate densities was especially evident under the cover that had been in place for a year (cover set in May 1979), it is interesting to note that sampling on July 24, 1980, 14 months after this cover had been set, showed approximately 4,100 invertebrates/sq m of bottom substrate. It is evident that the water under the cover did not become anaerobic and that invertebrates persisted, albeit in reduced numbers, under the cover.

Bulthuis (1984) tested clear, mesh, and black bottom covers for their effectiveness in eliminating the seagrass, *Heterozostera tasmanica*, in a coastal habitat in Australia. Although he did not determine the effect of the covers on the benthos beneath them, he found that after three months "there was no indication that the water became anaerobic under any of the barriers during the course of the study. Under all barriers, the surface of the sediments was always light colored and appeared oxidized." It is apparent that in this case, as in that of the lake in Wisconsin, water underneath the

barriers did not become anaerobic, despite the reduced water circulation and the decomposition of the vegetation beneath the covers. The Australian findings should be tempered, however, by the fact that the organic content of sediment in the coastal habitat is probably much less than that of the Potomac.

The type of bottom cover employed (permeable or impermeable), along with the physical and chemical features of the habitat, will be important in determining the dissolved oxygen levels and redox potentials of the water and sediment beneath the covers. The limited available studies do indicate that it is possible that water under the covers will not become anaerobic. In such a situation, infaunal invertebrates would probably be reduced but not eliminated. Since the plants beneath the covers would be eliminated, the epifaunal habitat previously available beneath the covers would no longer exist. However, infaunal and epifaunal invertebrates in neighboring areas (i.e., not under the covers), would be unaffected by their use.

It is interesting to note that the Aquascreen covers which Engel (1984) installed in May of 1980 had themselves become a colonization site for invertebrates. By August 1980, the upper surface of all of the covers were covered with chironomid larvae. Similarly, Mayer (1978) reported that the Aquascreen he used to control rooted aquatic plants in Chautauqua Lake, New York, became "covered with benthic fauna, especially fish food organisms."

## Dredging

Dredging can be used in an aquatic-plant-control capacity to accomplish a number of functions that eliminate aquatic plants directly or degrade the suitability of a habitat for producing them. These functions include: direct removal of aquatic plant standing crop; removal of root material contained in the sediment; removal of nutrient-rich sediments suitable for growing aquatic plants; and increasing the water depth to reduce the amount of light available at the bottom. A thorough discussion of the use of dredging techniques, hydraulically and with buckets or draglines, for lake restoration is presented by Peterson (1979). For aquatic-plant control in water bodies that cannot be completely drawn down, the only dredging technique widely used has been hydraulic dredging. Hydraulic dredging for control of submersed vegetation at sites in New York, Wisconsin, and British Columbia has been described in

published literature (George et al. 1982, Nichols 1984, Bryan 1978); however, none of these operations were for control of *Hydrilla*.

Technical aspects of planning and conducting a lake-restoration type of hydraulic dredging project have been described by Pierce (1970). Generally the largest problem associated with a hydraulic dredging operation is disposal of sediments; a nearby land-based site of adequate size must be located and secured. Additional problems may be encountered during operation if rocks and obstacles are within the area to be dredged and if strong winds occur (Bryan 1978).

Numerous environmental concerns and side effects are associated with dredging (Peterson 1979). Bryan (1978) observed that water-quality effects at the actual dredging site were minimal, consisting of a slight increase in turbidity and a slight reduction in dissolved oxygen levels immediately around the dredge. A generally greater environmental concern is likely to be associated with the design and operation of the disposal area.

The effectiveness of dredging for control of rooted submersed vegetation has generally been short lived. Bryan (1978) estimated that hydraulic dredging initially removed 90 to 95 percent of the Eurasian watermilfoil within treated areas but noted that such rapid milfoil reinfestation occurred, the duration of control was less than a year. In a review of techniques for controlling Eurasian watermilfoil, curlyleaf pondweed, and elodea, Nichols and Shaw (1983) noted that, in shallow water (i.e., within the photic zone), dredging "has little lasting impact on plant abundance." At sites where a significant depth increase has been effected by dredging (several meters), nuisance-level plant growth may be eliminated for a longer period of time (George et al. 1982, Nichols 1984), although the exact duration of control has yet to be determined by long-term studies.

Dredging for sediment removal and depth increase, used as an aquatic-plant-control technique, is recommended only for local control in the Potomac River due to its associated high cost, problems with securing and operating sediment disposal areas, operational difficulties, and probable short-lived control.

An alternative form of hydraulic dredging can be performed with a diver-operated dredge. This device, initially used for Eurasian watermilfoil control in British Columbia (Anonomous 1978), consists of a water pump driven by a small gasoline-powered engine, sections of flexible hose for intake and

discharge of sediments, and a catchment screen. The pump creates suction in the intake hose through a venturi connection, thus no dredged material is drawn through the pump. The intake hose is used by divers to vacuum up whole plants and plant-infested sediments. The dredged material is discharged onto a screen catchment basket that retains the plant material but allows the sediments to return to the water. The entire system is mounted on small pontoons and can be operated independent of the shore since no land-based sediment disposal operation is involved. A more detailed description of the original device may be found in Armour et al. (1980). Several smaller versions of the original device have been built and are being used in southern California to control *Hydrilla* in irrigation canals (McNabb, personal communication*). No published information is available on this operation.

Use of the diver-operated dredge is a very slow labor-intensive process. Killgore (1982) estimated that dredging rates for Eurasian watermilfoil in sandy substrate ranged from 0.005 to 0.015 acres/hour using the two-diver Canadian dredge. As such, the device is not recommended for attempting to control established infestations of rooted submersed vegetation; its most appropriate use is for eradication of small pioneer colonies of spreading aquatic plants (Newroth 1983). A distinct operational advantage of the diver-operated dredge is that it can be operated in areas containing rocks and obstacles (Anonomous 1978).

Because dredged sediments are returned to the water, there are distinct (albeit small, given limited pumping rates) water-quality effects associated with its operation. The resulting water-quality effects will of course be most dependent on the type of substrate being dredged. To date, use of the diver-operated dredge has largely been experimental; therefore, there are few data available on its environmental effects or the long-term effectiveness of its use.

Within the Potomac River, the diver-operated dredge may be a useful tool for controlling *Hydrilla* around docks and marinas, since no other mechanical technique could be used in an obstacle-filled environment.

---

* Terry McNabb, Aquatics Unlimited, Concord, Ga., 1985.

## Sediment Agitation

Mechanical agitation of plant-infested sediments has been used experimentally to achieve some control of Eurasian watermilfoil in British Columbia (Newroth 1983). Sediments are mechanically agitated by any of several means, causing plant material to become dislodged from the sediment and float. The floating material collects against fragment barriers (see next section) and is removed following completion of the operation.

Equipment design and operational aspects of this technique are discussed by Bryan and Armour (1982). Sediments can be agitated using a conventional agricultural cultivator or an actively driven rotavator. The agitation device may be propelled by any of several types of equipment, depending upon the water depth. In very shallow waters, a conventional tractor can be used; a tracked amphibious vehicle can be used in slightly deeper waters; and an aquatic plant harvester can be used in deep waters. Operations should be conducted when plant biomass is low (i.e., in fall, winter, or spring), in order to reduce the amount of collected material that must be handled and to reduce the number of viable fragments that may be released (Newroth 1983). Use of this technique is of course restricted to areas that are free from obstructions and objects that may damage or be damaged by the equipment.

Bryan and Armour (1982) estimated that 90 percent of the plant material within a test area was initially removed by active rotavating, but noted that reinfestation was rapid. Initial removal of plant material is greatest in loose unconsolidated sediments; removal decreases as sediments become more consolidated and cohesive.

Use of this technique may have certain water-quality and environmental consequences. Bryan and Armour (1982) noted a dramatic but relatively short-lived increase in turbidity following rotavating. The type and degree of environmental effects would expectably depend most on the composition and benthic inhabitants of the sediment being agitated.

This technique is largely experimental and has not been used in *Hydrilla*. Its effectiveness would be dependent on removing tubers from the sediment. This seems unlikely since tubers are negatively buoyant. Prior to maturation, tubers are attached to the roots and indirectly to the buoyant stems. Agitation operations conducted prior to detachment of the tubers would create a great deal of drifting plant material and would likely mechanically detach

many of the tubers, decreasing effectiveness. Additionally, the agitation would impact benthic inhabitants and might cause water-quality problems. For these reasons, this technique is not recommended for control of *Hydrilla* within the Potomac.

## Confinement and Collection of Detached Plant Fragments

Dispersal of buoyant plant fragments is one of the primary means by which *Hydrilla* infestations spread (see chapter on ecology). Measures should be taken to prevent dispersal of fragments generated by any type of control operation. Techniques for confining or collecting plant fragments can be either active or passive.

Active collection can be performed using a pusher boat, described by Smith (1980). This device consists of a small flat-bottom boat powered with a long-shaft outboard motor. A pusher system mounted on the bow consists of an expanded metal rake that can be raised and lowered by an electric motor. The pusher boat is used for in-water transport of large quantities of detached plant material, such as may be generated by cutting operations.

Passive confinement or collection can be performed by fragment barriers that consist of netting material suspended in the water from a floating boom. Temporarily placed fragment barriers have been used in British Columbia to confine areas in which Eurasian watermilfoil was being harvested or dredged (Newroth 1979, Bryan and Armour 1982). The barrier design consisted of 1.2-cm-mesh nylon netting approximately 2-m wide suspended in the water on a lead line from cylindrical floats. Following mechanical treatment, the barrier must be cleaned and removed.

Longer term installation of fragment barriers has been used in the Okanogan River in Washington State and British Columbia in an attempt to stop the downstream spread of Eurasian watermilfoil (Newroth 1979, Dardeau and Lazor 1982). The barriers were installed at strategic constrictions in the river and cleared of collected vegetation several times per week. Barrier design differed slightly from those of temporary placements in that the meshes were large, up to 5 cm, and were made of more rigid material such as wire mesh (Newroth 1979). Dardeau and Lazor (1982) estimated that the overall collection efficiency of the surface-suspended barriers was only in the range of 23.6 to 86.1 percent, even though the "overwhelming majority" of the drifting

material was within the 0- to 1-ft depth range. This would suggest that, in the prevention mode, this barrier design can slow the infestation spread, but cannot stop it. An additional disadvantage of using this type of barrier in a flowing water system is that it must contend with large floating debris that could result in structural damage to the barrier.

Fragment barriers were installed in the Dike Marsh area of the Potomac River for the National Park Service during the spring of 1984 to prevent the movement of vegetatively active *Hydrilla* stems and turions from this heavily infested area. Results were less than favorable; the full-water-column fine-mesh (0.5 mm) barrier became so clogged with sediment that seams were damaged, ending its usefulness. James (personal communication*) indicated that better results may be obtained if a larger mesh size net is used to compensate for high silt loading and if the barrier is only placed at the surface (1 to 2 ft below water surface).

Short-term installation of fragment barriers within the Potomac River would probably be the most appropriate confinement and collection technique for further use. Active collection techniques would not be needed since cutting without harvesting is not recommended, and active collection of large mats of detached drifting *Hydrilla* observed during the fall (Allari, unpublished data**) would be futile (Steward, personal communication†). Long-term installation of fragment barriers to prevent spreading of the infestation may have application at some locations although probably not many, since the status of the infestation is well beyond that which may be helped by a prevention technique.

## Conclusions and Recommendations

The significance of the environmental impacts associated with the application of the control techniques discussed is largely dependent upon the relative amount of vegetated area treated. When a relatively small portion is treated, overall effects are likely to be small and have little detrimental impact on the system; when a relatively large portion is treated, effects are

---

* Russell I. James, Ecoscience, Inc., Old Forge, Pa., 1985.
** Ruth Allari, North Virginia Community College, Alexandria, Va., 1984.
† Kerry Steward, US Department of Agriculture, Ft. Lauderdale, Fla., 1984.

likely to be more significant. For this reason, it is recommended that treatment be limited to the minimum area required to meet the demands of river users.

Cutting and harvesting does not appear to have the potential to achieve more than temporary control of monoecious *Hydrilla*; however, it can be used effectively to achieve immediate control from nuisance-level infestations. Cutting alone (i.e., without collection) is not recommended since it would release viable *Hydrilla* fragments that could become established in other downstream sites.

Research is needed to develop capabilities to predict the growth rate of monoecious *Hydrilla* within the Potomac River. The analytical capability would be used to assist planners in determining the optimum time for harvesting.

Effects of cutting and harvesting on plant-species composition within multispecies plant beds are not known. Studies should be initiated to determine such effects.

Direct chemical and physical effects of a harvester's operation on the aquatic environment are minimal.

Removal of a portion of the macroinvertebrates and fish while harvesting a plant-infested site is unavoidable. However, if the treated area is relatively small compared to the total vegetated area, then effects on macroinvertebrates and fish communities should be minimal. The spawning activities of important fish species should be considered in developing the harvesting schedule, and harvesting should not be conducted at critical spawning sites during times of known spawning activity.

Contracting for harvesting services may be preferable to performing the operation in-house since contracting allows for greater flexibility in meeting variable demand for harvesting and it avoids the necessity of finding skilled seasonal employees to operate the equipment.

Placement of bottom-covering fabrics and membranes may be a useful technique for controlling *Hydrilla*. Studies initiated in 1984 with the placement of two types of material will be continued in order to determine what types of material (and what mechanisms of control) are most effective against *Hydrilla*. As part of these studies, factors limiting the duration of control effected by the material will be examined, and in-situ procedures for extending the effectiveness of the material will be tested.

If it is determined that bottom-covering fabrics or membranes effectively control *Hydrilla* and if large area applications are considered, then additional research should be conducted in the following areas: (a) methods for rapid placement of the material; (b) sediment and water-quality impacts of placement of the material; and (c) impacts of material on macroinvertebrates and the fish that feed upon macroinvertebrates.

Conventional dredging techniques (hydraulic, bucket, or dragline) are recommended only for localized control of aquatic plants in the Potomac River because of high cost, required sediment disposal sites, and questionable effectiveness.

Use of a diver-operated dredge may be a useful technique for controlling *Hydrilla* growth around pilings at docks and marinas. Field tests will be conducted during the 1985 growing season.

If the objective of a management program is to minimize the spread of *Hydrilla* to uninfested areas, then measures should be taken to minimize the dispersal of plant fragments created by any type of control operation. When necessary, a fragment barrier should be installed temporarily.

## Contributors

| | |
|---|---|
| Mr. Bruce M. Sabol* | US Army Engineer Waterways Experiment Station<br>ATTN: WESEN-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. David C. Beckett | US Army Engineer Waterways Experiment Station<br>ATTN: WESER-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. Donald A. Graetz | Department of Soil Science<br>University of Florida<br>Gainesville, FL |
| Mr. Russell I. James | Ecoscience Inc.<br>Old Forge, PA |
| Mr. Terry McNabb | Aquatics Unlimited<br>Concord, GA |

* Mr. Sabol was primarily responsible for the development of this chapter and should be used as a point of contact regarding its content.

## References


Anonymous. 1978. Information bulletin-aquatic plant management program, Okanagan Lakes volume IV: a review of mechanical devices used in the control of Eurasian Watermilfoil in British Columbia. Water Investigation Branch, Ministry of the Environment, Province of British Columbia, Victoria.

Armour, G. D., D. W. Brown, and K. T. Marsden. 1979. Studies on aquatic macrophytes, part XV: an evaluation of bottom barriers for control of Eurasian Watermilfoil in British Columbia. Waters Investigations Branch, Ministry of Environment, Province of British Columbia, Canada.

Armour, G. D., R. S. Hanna, I. P. Walters, and M. D. Maxnuk. 1980. Studies on aquatic macrophytes, part XIV: summary of mechanical aquatic plant management. Inventory and Engineering Branch, Ministry of Environment, Province of British Columbia, Victoria.

Bartell, S. M., and J. E. Breck. 1979. Simulated impact of macrophyte harvesting on pelagic phosphorus cycling in Lake Wingra, p. 229-250. In J. E. Breck, R. T. Prentki, and O. L. Loucks [ed.], Aquatic plants, lake management, and ecosystem consequences of lake harvesting, proceedings of conference at Madison, Wisconsin, February 14-16, 1979. Institute for Environmental Studies, University of Wisconsin, Madison.

Boston, H. L., and M. A. Perkins. 1982. Water column impacts of macrophyte decomposition beneath fiberglass screens. Aquatic Botany 14:15-27.

Boyle, T. P. 1979. Responses of experimental lentic aquatic ecosystems to suppression of rooted macrophytes, p. 269-284. In J. E. Breck, R. T. Prentki, and O. L. Loucks [ed.], Aquatic plants, lake management, and ecosystem consequences of lake harvesting, proceedings of conference at Madison, Wisconsin, February 14-16, 1979. Institute for Environmental Studies, University of Wisconsin, Madison.

Breck, J. E., and J. F. Kitchell. 1979. Effects of macrophyte harvesting on simulated predator-prey interactions, p. 211-228. In J. E. Breck, R. T. Prentki, and O. L. Loucks [ed.], Aquatic plants, lake management, and ecosystem consequences of lake harvesting, proceedings of conference at Madison, Wisconsin, February 14-16, 1979. Institute for Environmental Studies, University of Wisconsin, Madison.

Bryan, A. D. (ed.) 1978. Studies on aquatic macrophytes, part VIII, experimental hydraulic dredging for aquatic weed control in Vernon Arm, Okanagan Lake, 1975. Water Investigations Branch, Ministry of the Environment, Province of British Columbia, Victoria.

Bryan, A. D., and G. D. Armour. 1982. Studies on aquatic macrophytes, part XI: rotavating and jetting for aquatic weed control in Okanagan and Kalamalka Lakes. Assessment and Planning Division, Ministry of Environment, Province of British Columbia, Victoria.

Bulthuis, D. A. 1984. Control of the seagrass *Heterozostera tasmanica* by benthic screens. J. Aquat Plant Manage. 22: 41-43.

Burton, T. M., D. L. King, and J. L. Ervin. 1979. Aquatic plant harvesting as a lake restoration technique, p. 177-185. In Lake restoration-proceedings of a national conference, August 22-24, 1978, Minneapolis, Minnesota, United States Environmental Protection Agency, Washington, D.C.

Byrnes, B. H., D. R. Keeney and D. A. Graetz. 1972. Release of ammonium-N from sediments to waters, p. 249-259 In. Proc. 15th Conf. Great Lakes Research 1972.

Canellos, G. 1981. Aquatic plants and mechanical methods for their control. The Mitre Corporation, McLean, Virginia. U.S. Environmental Protection Agency contract no. 68-01-5965.

Carpenter, S. R. 1976. Some environmental impacts of mechanical harvesting of nuisance submerged vascular plants. Masters thesis, University of Wisconsin, Madison. 95 p.

Carpenter, S. R., and M. S. Adams. 1977. Environmental impacts of mechanical harvesting of submerged vascular plants. Institute for Environmental Studies. University of Wisconsin, Madison.

Carpenter, S. R., and A. Gasith. 1978. Mechanical cutting of submersed macrophytes: immediate effects on littoral water chemistry and metabolism. Water Research 12: 55-57.

Cooke, G. D. 1980. Covering bottom sediments as a lake restoration technique. Water Resources Bulletin 16(5): 921-926.

Cooke, G. D., and M. E. Gorman. 1980. Effectiveness of DuPont typhar sheeting in controlling macrophyte regrowth after overwinter drawdown. Water Resources Bulletin 16(2): 353-355.

Crowder, L. B., and W. E. Cooper. 1979. The effects of macrophyte removal on the feeding efficiency and growth of sunfishes: evidence from pond studies, p. 251-268. In J. E. Breck, R. T. Prentki, and O. L. Loucks [ed.], Aquatic plants, lake management, and ecosystem consequences of lake harvesting, proceedings of conference at Madison, Wisconsin, February 14-16, 1979. Institute for Environmental Studies, University of Wisconsin, Madison.

Dardeau, E. A., and R. L. Lazor. 1982. Implementation of the large-scale operations management test in the state of Washington. Miscellaneous Paper A-82-7, U. S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Diaz, R. J., and D. F. Boesch. 1977. Habitat development field investigations, Windmill Point Marsh development site, James River, Virginia; Appendix C: environmental impacts of marsh development with dredged materials: acute impacts on the macrobenthic community. Technical

Report D-77-23, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Dresler, P. V. 1980. The Asiatic clam, *Corbicula fluminea* (Muller), in tidal Potomac River, Maryland. Estuaries 3:150-151.

Engel, S. 1984. Evaluating stationary blankets and removable screens for macrophyte control in lakes. J. Aquat. Plant Manage. 22: 43-48.

Engel, S., and S. A. Nichols. 1984. Lake sediment alteration for macrophyte control. J. Aquat. Plant Manage. 22: 38-41.

George, C. J., P. L. Tobiessen, P. D. Snow, and T. K. Jewell. 1982. The restorational dredging of Collins Lake, Scotia, New York. EPA project summary EPA-600/53-81-017, U. S. Environmental Protection Agency, Washington, D.C.

Graetz, D. A., D. R. Keeney, and R. A. Aspiras. 1973. Eh status of lake sediment-water systems in relation to nitrogen transformations. Limnol. and Oceanog. 18: 908-917.

Haller, W. T., J. V. Shireman, and D. F. DuRant. 1980. Fish harvest resulting from mechanical control of hydrilla. Transactions of the American Fisheries Society 109: 517-520.

Hasler, A. D. 1969. Cultural eutrophication is reversible. Bioscience, 19(5): 425-431.

Jones, G. B., and G. D. Cooke. 1983. Burlap screening to control macrophyte growth. Lake Line 3(1): 6-7.

Jorgensen, B. B. 1983. Processes at the sediment-water interface. In B. Bolin and R. B. Cook, [ed.], Major biogeochemical cycles and their interactions. Wiley and Sons, New York.

Killgore, K. J. 1982. Large-scale operations management test of prevention methodologies: milfoil fragment and diver dredge studies, p. 258-265. In Proceedings 16th annual meeting aquatic plant control research planning and operations review. 17-19 November 1981, St. Paul, Minnesota. Miscellaneous Paper A-82-3, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Kimbel, J. C., and S. R. Carpenter. 1979. The dynamics of *Myriophyllum spicatum* biomass following harvest, p. 43-50 In J. E. Breck, R. T. Prentki, and O. L. Loucks [ed.], Aquatic plants, lake management, and ecosystem consequences of lake harvesting, proceedings of conference at Madison, Wisconsin, February 14-16, 1979. Institute for Environmental Studies, University of Wisconsin, Madison.

Lewis, D. H., I. Wile, and D. S. Painter. 1983. Evaluation of terratrack and aquascreen for control of aquatic macrophytes. J. Aquat. Plant Manage. 21: 103-104.

Livermore, D. F. 1954. Harvesting underwater weeds. Waterworks Engineering 107(2): 118-120.

Livermore, D. F., and R. G. Koegel. 1979. Mechanical harvesting of aquatic plants: an assessment of the state of the art, p. 307-328. In J. E. Breck, R. T. Prentki, and O. L. Loucks [ed.], Aquatic plants, lake management, and ecosystem consequences of lake harvesting, proceedings of conference at Madison, Wisconsin, February 14-16, 1979. Institute for Environmental Studies, University of Wisconsin, Madison.

Livermore, D. F., and W. E. Wunderlich. 1969. Mechanical removal of organic production waterways, p. 494-519. In J. E. Breck, R. T. Prentki, and O. L. Loucks [ed.], Aquatic plants, lake management, and ecosystem consequences. Institute for Environmental Studies, University of Wisconsin, Madison.

Mayer, R. J. 1978. Aquatic weed management by benthic semibarriers. J. Aquat. Plant Manage. 16: 31-33.

Mikol, G. F. 1984. Effects of mechanical control of aquatic vegetation on biomass, regrowth rates and juvenile fish populations in Saratoga Lake, New York, p. 456-462. In Lake and reservoir management-proceedings of the (3rd) annual conference of The North American Lake Management Society, October 18-20, 1983, Knoxville, Tennessee.

Newroth, P. R. 1974. Studies on aquatic macrophytes, part V: aspects of aquatic weed control by mechanical harvester 1972-1973. Ministry of Environment, Province of British Columbia, Victoria.

Newroth, P. R. 1979. British Columbia aquatic plant management program, J. Aquat. Plant Manage. 17: 12-19.

Newroth, P. R. 1983. Mechanical control of Eurasian watermilfoil. Lake Line 3(2) 8-9.

Nichols, S. A. 1973. The effects of harvesting aquatic macrophytes on algae. Wisconsin Academy of Sciences, Arts, and Letters 61: 165-172.

Nichols, S. A. 1984. Macrophyte community dynamics in a dredged Wisconsin lake. Water Resources Bulletin 20(4): 573-576.

Nichols, S. A., and B. H. Shaw. 1983. Review of management tactics for integrated aquatic weed management of Eurasian watermilfoil (*Myriophyllum spicatum*), curly leaf pondweed (*Potamogeton crispus*), and elodea (*Elodea canadensis*), p. 181-192. In Lake restoration, protection and management-proceedings of the (2nd) annual conference of the North American Lake Management Society, October 26-29, 1982, Vancouver, British Columbia.

Pearson, R. G., and N. V. Jones. 1978. The effects of weed-cutting on the macro-invertebrate fauna of a canalised section of the River Hull, a northern English chalk stream. Journal of Environmental Management 7:91-97.

Perkins, M. A. 1984. An evaluation of pigmented nylon film for use in aquatic plant management, p. 467-472. In Lake and reservoir management-proceedings of the (3rd) annual conference of the North American Lake Management Society, October 18-20, 1983, Knoxville, Tennessee.

Perkins, M. A., H. L. Boston, and E. F. Curren. 1980. The use of fiberglass screens for control of Eurasian watermilfoil. J. Aquat. Plant Manage. 18: 13-19.

Peterson, S. A., W. L. Smith, and K. W. Maleug. 1974. Full-scale harvest of aquatic plants: nutrient removal from a eutrophic lake. Journal WPCF 46: 697-707.

Peterson, S. A. 1979. Dredging and lake restoration, p. 105-114. In Lake restoration-proceedings of a national conference, August 22-24, 1978, Minneapolis, Minnesota. United States Environmental Protection Agency, Washington, D. C.

Pierce, N. D. 1970. Inland lake dredging evaluation. Wisconsin Department of Natural Resources, Madison.

Pullman, D., and J. Craig. 1981. Pond bottom liner strategy is more than a coverup. Weeds Trees and Turf 20: 36-39.

Sabol, B. M. 1983. Simulated mechanical control of aquatic plants in Buffalo Lake, Wisconsin. Miscellaneous Paper A-83-8, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Sabol, B. M., and T. D. Hutto. 1984. Simulating mechanical control operations. Information Exchange Bulletin Vol A-84-1, U. S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Miss.

Seitzinger, S. P., S. W. Nixon, and M. E. Q. Pilson. 1984. Denitrification and nitrous oxide production in a coastal marine ecosystem. Limnol. Oceanog. 29: 73-83.

Smith, G. N. 1979. Recent case studies of macrophyte harvesting costs: options by which to lower cost, p. 345-356. In J. E. Breck, R. T. Prentki, and O. L. Loucks [ed.], Aquatic plants, lake management, and ecosystem consequences of lake harvesting, proceedings of conference at Madison, Wisconsin, February 14-16, 1979. Institute for Environmental Studies, University of Wisconsin, Madison.

Smith, P. A. 1980. Mechanical harvesting of aquatic plants; report 2, evaluation of selected handling functions of mechanical control. Technical Report A-78-3, U. S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Miss.

Steward, K. K. 1970. Nutrient removal potentials of various aquatic plants. Hya. Control J. 8(2): 34-35.

Swales, S. 1982. Impacts of weed-cutting on fisheries: an experimental study in a small lowland river. Fish Mgmt. 13(4): 125-137.

Titus, J., R. A. Goldstein, M. S. Adams, J. B. Mankin, R. V. O'Neill, P. R. Weiler, Jr., H. H. Shugart, and R. S. Booth. 1975. A production model for *Myriophyllum spicatum* L., Ecology 56(5): 1129-1138.

Wile, I. 1975. Lake restoration through mechanical harvesting of aquatic vegetation. Verh. Internat. Verein. Limnol. 19: 660-671.

Wile, I. 1978. Environmental effects of mechanical harvesting. J. Aquat. Plant Manage. 16: 14-20.

Zisette, R. 1983. Milfoil control evaluation in the King County/Seattle region. Tech Memorandum, Metro Water Quality Division, Washington. 24 pp.

# CHAPTER V: CHEMICAL CONTROL OF *HYDRILLA*

## Overview

The objective of this section is to identify those herbicides that are known to be effective in managing *Hydrilla*. Only herbicides that are registered for aquatic use under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) are considered. These herbicides are copper complexes, diquat, endothall, and fluridone. Only diquat and copper complexes have an EPA-approved label for use in moving water. This section summarizes background information on those chemical and physical characteristics of the Potomac River that may influence chemical weed control, the environmental fate of these herbicides in aquatic environments, toxicity of these chemicals to nontarget organisms, and herbicide effects on *Hydrilla* shoots and reproductive propagules. Conclusions and recommendations are presented, and a list of additional research needs to assist in determining which herbicides are most beneficial for use in the Potomac River is provided.

## Herbicide Fate in the Aquatic Environment

The fate, effectiveness, and impact of herbicides in aquatic environments depend on the following characteristics of the habitat: (1) area and depth, (2) water movement, (3) presence of aquatic vegetation, (4) amount of mud or suspended silt, (5) nature of rock and soil environment, (6) hardness and pH value, (7) amount of sunlight, and (8) presence of fish, fauna, and microorganisms (Coats et al. 1964, Summers 1980). In the tidal zone of the Potomac River, many of these characteristics have to be considered before using herbicides to control *Hydrilla*. Tidal currents as high as 0.5 m/sec occur in this region; hence, herbicides may be potentially distributed over a wide area of the river. The chemistry of these waters, including hardness, conductivity, and particulate and dissolved organic matter content, provides a variety of complexing, ion-exchange, and other adsorption reactions in the water column. The potential for transport of herbicides from the area of application and herbicide dilution must be determined in relation to any negative impacts to native submersed grasses that have begun to revegetate this region of the Potomac River. Also, the potential wide distribution of herbicides must be

evaluated as related to the economically important fisheries that may utilize the tidal river (e.g., striped bass).

Table 1 is a list of recommended aquatic herbicides that are safe and effective for controlling *Hydrilla*.

Copper ion complexes

Copper characteristics and use. Copper has been used for many years for algae control and more recently to manage *Hydrilla*. The toxicology of copper is relatively well understood, and copper complexes have gained recent acceptance due to minimal environmental impact (Table 2). There are currently no restrictions on the use of treated water. The water may be used for domestic purposes, swimming, fishing, and irrigation immediately after treatment with copper. Although copper sulfate may be used in aquatic environments for algae control, only copper complexes that are registered for control will be considered in the following discussion. Cutrine-Plus, K-TEA Algaecide, Koplex, and Komeen are registered for aquatic use.

a. Cutrine-Plus. Cutrine-Plus is a liquid copper ethanolamine complex herbicide manufactured by Applied Biochemists, Inc. (Applied Biochemists 1983). Doses of 0.4 to 1.0 mg/ℓ will control *Hydrilla*, and a 3-hr contact time is required in lotic (flowing) waters. Some states require a permit when Cutrine-Plus is used in public water.

Toxicity of Cutrine-Plus is related to water hardness. Cutrine-Plus is generally nontoxic to fish and wildlife at recommended dosages. Concentrations that are acutely toxic to fish in moderately hard and hard waters are a factor of two or less of the recommended dose for *Hydrilla*. Trout, tropical fish, ornamental goldfish, and other sensitive fish may be adversely affected in soft waters with less than 50 mg/ℓ as $CaCO_3$ hardness. Fish may be caught and consumed immediately after Cutrine-Plus application.

Cutrine-Plus requires light and water temperatures above 15°C to be effective. Its effects on the target species may be observed in seven to ten days after treatment. Retreatment may be necessary to obtain the desired control level. Apparently Cutrine-Plus is more effective on young, actively growing submersed vegetation. Cutrine-Plus may be sprayed or injected and is compatible in a tank mix with diquat and endothall.

b. Komeen and Koplex. Komeen and Koplex are copper-ethylenediamine complex herbicides manufactured by Sandoz, Inc., and Kocide Chemical Corporation, respectively. Komeen is normally applied at 57 to 151 ℓ/surface hectare (6

to 16 gal/surface acre) to control *Hydrilla*. The site-specific recommended dose is based on water volume, not surface acreage. Komeen is a liquid that may either be sprayed or injected below the water surface. Nalquatic is a recommended adjuvant for spraying *Hydrilla*. Koplex has similar use properties and considerations. Effects of Koplex may be seen in 3 to 6 days with full effects manifested in 4 to 6 weeks. Komeen and Koplex may be inverted with an adjuvant or used in combination with diquat. Both water hardness and rapid decay of treated vegetation must be considered prior to its use.

c. K-TEA Algaecide. K-TEA Algaecide is a copper-triethanolamine complex herbicide registered for aquatic use. It is registered for use in Florida for control of *Hydrilla*. K-TEA Algaecide has similar properties to the other copper-ethanolamine complexes. If K-TEA Algaecide is used in water with a pH of 6 or less, the copper chelate may be broken and subsequently the copper ion may be precipitated.

Fate processes and rate constants. When introduced to aquatic systems, copper may persist in the sediment for long periods (Table 3). Only the organics used to complex the copper ion are subject to biodegradation. Copper complexes are not subject to photolysis or volatilization. The water's pH and hardness may be factors that regulate the persistence of copper complexes and their bioavailability. Copper will bioconcentrate but probably not above FDA action levels* in herbicide treatment situations. Copper will also sorb significantly to organic sediments and clays; copper ions form insoluble copper hydroxides, phosphates, and carbonates in water with pH >7. Copper complexes are soluble and are mobile when introduced to flowing waters.

Compartmentalization and persistence. Copper complexes used to control *Hydrilla* in the Potomac River would probably be subjected to rapid dilution. Copper concentrations in aquatic vegetation and surficial sediments may measurably increase after introduction and repeated use may result in adverse effects on nontarget species.

There is relatively little published information on the stability of copper complexes in aquatic systems. Particularly, there is a need for information on degradation of the organic complexing agents. Most of the

---

* FDA action level is the chemical concentration in the edible tissue of an organism that upon exceedence, renders the organism unsatisfactory for human consumption.

bioavailability and bioconcentration data are for copper ion and not copper complexes. Accurate predictions of copper-complex compartmentalization and persistence would require information on these processes if the behavior of copper complexes is significantly different from that of the copper ion.

Diquat

Characteristics and use. Diquat is a dibromide salt of the dipyridylium class of chemicals that is ionic when in aqueous solutions (Crafts 1975). Diquat is a contact herbicide with rapid desiccant action (Table 4). Diquat generates free radical action in plants causing rapid disruption of cellular functions (Ashton and Crafts 1981). Diquat has been used for about twenty years and is registered as an aquatic herbicide. The double positively charged diquat cation affords this herbicide some unique properties.

Generally, treated water cannot be used for animal consumption, swimming, or spraying within 10 days after treatment unless analysis shows that the water does not contain more than 0.01 mg/ℓ diquat ion. Treated water may not be used for drinking purposes or overhead irrigation until 14 days after treatment or unless an approved analysis shows that the water does not contain more than 0.01 mg/ℓ diquat ion. The interim tolerance for diquat in potable waters is 0.01 mg/ℓ. There is no restriction on the catching and removal of fish from the treated area. If a commercial fish-processing industry is located on the shoreline and uses water from the river, then diquat must not be applied near the intake. The registration label also specifies that diquat is not to be used in muddy water.

Diquat may be used in tank mixes with copper complexes for control of *Hydrilla*. Inverts may also be used and applied with weighted hoses. Initial concentrations of 1 to 2 mg/ℓ are recommended for *Hydrilla* control. Barrett (1981) stated that a minimum contact time of 24 hr was required at the recommended treatment rate of 1 mg/ℓ diquat. Mackenzie (1968) observed that control of hydrilla was obtained with diquat at 0.5-1.0 mg/ℓ only where the water was static and where heavy rainfall did not dilute the treatment within 48 hr after application.

Of the numerous diquat formulations marketed, only Diquat Water Weed Killer manufactured by Chevron-Ortho is registered for *Hydrilla* control (Table 1). This liquid herbicide is 35.3 percent by weight diquat dibromide and contains 0.24 kg diquat cation/ℓ. Diquat Water Weed Killer is recommended for control of *Hydrilla* in quiescent or slowly moving water. Use of 18.7 ℓ/

surface hectare with bottom placement and Nalquatic, an adjuvant manufactured by Nalco Chemical Company, is also recommended by Chevron on the registration label. There are special cautions against use of Diquat Water Weed Killer in muddy water or on silt-covered plants.

Diquat Water Weed Killer acts relatively rapidly and oxygen consumption due to vegetation decay may be of concern. The manufacturer recommends that when a large area is to be treated, the area should be subdivided and treated sequentially several weeks apart to minimize oxygen depletion in the water column during plant decomposition.

Fate processes and rate constants. Diquat is relatively water soluble and is subject to dilution and dispersion in aquatic systems (Table 5). Diquat is unlikely to bioconcentrate significantly or to persist in the tissues of aquatic organisms under normal use (Haven 1969). Volatilization and oxidation of diquat are not significant removal processes. Alkaline hydrolysis may contribute to transformation of diquat, but the importance of this process is not well known. Similarly, photolysis may contribute to altering the chemical structure of diquat in shallow waters resulting in reduced efficacy, but the magnitude of this process in nature is not well known. Biotransformation of diquat is also a poorly understood process from the perspective of environmentally relevant rates. Diquat biotransformation may be significantly altered when the herbicide is sorbed to particulates (Simsiman and Chesters 1976). Diquat adsorbed on clays is also not bioavailable. Its herbicidal properties depend upon characteristics of absorbents.

Articles have been included (Encl. 1-4, Appendix A) that clarify the potential for ethylene dibromide (EDB) exposure to the public and the environment through diquat usage for aquatic plant management. Florida has established a 0.1-$\mu g/\ell$ (ppb) EDB tolerance in water, which is the minimum detectable level. At no time will 0.1 $\mu g/\ell$ (ppb) EDB be exceeded through the use of diquat for aquatic plant control operations, since existing formulations contain a maximum of 10 ppm EDB. Hence, no adverse environmental or public risk should be expected through the use of diquat according to label instructions.

Compartmentalization and persistence. Sediment sorption is apparently very important, but systematic information that would allow accurate predictions of sorption is lacking (Coats et al. 1966). Shortly after introduction to an aqueous environment, diquat is found strongly sorbed to suspended clay

particles (Poinke and Chesters 1973, Summers 1980). Calculated sorption constants, Kp's, from available data indicate that reasonable values range from 10 to 50 (Narine and Grey 1982). The herbicidal properties of diquat are inactivated for an indeterminant period by clay and suspended sediment. The silt and clay contents of sediment may influence the persistence of diquat (Coats et al. 1966, Summers 1980). Dissolved organic carbon may also play a role in diquat persistence by complexing (Khan 1974); however, its herbicidal activity would not be significantly changed. The kind of cations present in the absorbent and active sorption sites will regulate adsorption of bipyridylium herbicides.

Data from a few studies indicate that diquat half-life may range from 1 or 2 days to several days. Reported overall transformation and/or removal rates for diquat range from 0.04 to 0.925 per day.

Endothall

Characteristics and use. Endothall has been registered as an aquatic herbicide since 1960 and is commonly applied as the dipotassium salt for aquatic weed management. Unlike many herbicides, endothall contains only carbon, hydrogen, and oxygen (Table 6). With a molecular formula of $C_8H_{10}O_5$, endothall is an odorless crystalline white solid. Rather than being applied as the free acid, endothall is converted to its inorganic or amine salts. These salts are then applied for aquatic use as aqueous concentrates or granules.

Although endothall was originally developed as an agricultural herbicide, it has been shown to be effective for the control of many aquatic weeds. It is now used extensively not only in ponds and lakes, but also in irrigation and drainage channels (Blackburn and Weldon 1964).

Various workers have shown that not only the timing of herbicide application is important in the chemical control of aquatic plants, but also the water temperature and the age of the target plants. Walker (1963) showed that aquatic vegetation was most susceptible to control by endothall applied when the water temperature was above 15°C and when the vegetation is young and actively growing. As the plants reached maturity and infestation became more dense, higher rates of application were required to kill the stands.

Endothall is a typical contact-type membrane-active compound that causes a general molecular disorder in the plasmalemma and other cell membranes, possibly by ionic interactions. The general disruption of the normal

compartmentalization within the cells causes abnormal flows of substrates and metabolites and an altered pattern of enzyme activity (Maestri 1967, Thomas 1966).

Though the use of endothall-based herbicides to control aquatic weeds is common, few studies have examined the effects of herbicidal treatment on the surrounding waters. The release of compounds from the herbicide-affected plants presents the possibility of altering the chemical composition of the water system (Westerdahl 1981). The literature is often contradictory in regard to these effects. Walker (1963) found increases in the concentrations of phosphorus and nitrogen compounds, calcium, magnesium, and potassium in the water of entire and partitioned areas of ponds treated with endothall compounds. In contrast, Holmberg (1973) did not report a change in any of these parameters in the water of a whole pond that he examined after endothall treatment to control milfoil. Westerdahl (1981) reported no change in nitrogen and phosphorus concentrations after localized treatment in Gatun Lake, Panama, to control *Hydrilla*; he attributed the lack of change to algal utilization of any released nutrients. Rodgers et al. (1984) did not observe any effects of endothall on nontarget species or water quality when recreational areas were treated in a reservoir.

Holmberg (1973) reported significant changes in the planktonic chlorophyll a concentrations in pond water after Aquathol treatment. Though when localized treatment was utilized in a lake, no significant alteration of the chlorophyll a concentrations was found (Rodgers et al. 1984).

The environmental parameter most often examined in conjunction with aquatic herbicide treatments is the dissolved oxygen concentration in the water. Again, inconsistent findings are present in the literature. Strange and Schreck (1976), Kilgore (1981), and Westerdahl (1981) report significant alterations in the dissolved oxygen concentrations at localized areas treated with endothall. Holmberg (1973) and Rodgers et al. (1984) did not show major alteration in the dissolved oxygen levels that were attributable to herbicidal treatment.

Endothall is manufactured as an aquatic herbicide by the Pennwalt Corporation (Table 1). Both liquid (Aquathol K) and granular (Aquathol) formulations of the dipotassium salt are registered for aquatic use. Other alkylamine salts of endothall are registered for *Hydrilla* control in Florida, Georgia, Texas, and Alabama.

Aquathol K, the dipotassium salt liquid formulation of endothall, is widely used for *Hydrilla* control. Aquathol K is also used with copper complexes in tank mixes for *Hydrilla* control. In flowing water, the minimum contact time required for optimum control of submersed weeds is 2 hr (Pennwalt 1984). Water-use restrictions for Aquathol K include 24 hr for swimming, 5 days for consumption of fish from treated water, 7 days for potable water use, and 7 days for irrigation of grass and nonfood crops. The tolerance for endothall in drinking water supplies is currently under negotiation. The definition of "treated water" under the consumption of fish restriction is purposely ill-defined, since boundaries would be impossible to identify and enforce because of herbicide dispersion. Consequently, the EPA is most concerned about having the responsible agencies or applicators inform the public of the treatment and post the treated areas with signs listing the aforementioned restrictions.

Aquathol Granular Aquatic Herbicide is also registered for aquatic use and is used for *Hydrilla* control. The water-use restrictions for Aquathol Granular are similar to those for Aquathol K.

Compartmentalization and persistence. Endothall compartmentalization and persistence has been thoroughly studied (Table 7). The primary pathway for endothall dissipation is biotransformation. Endothall does not sorb appreciably to sediments nor does it significantly bioconcentrate in biota. The overall half-life for the dipotassium salt of endothall in aquatic systems is 1 to 4 days. Since biotransformation is the major pathway for removal of endothall and endothall is readily mineralized, the biotransformation rate must be well known for accurate prediction of persistence in any specific aquatic system. Horizontal dispersion and dilution may also be important factors regulating endothall concentrations at a site. Endothall is rapidly released from the granular formulation upon introduction to aquatic systems, and the formulation does not apparently affect persistence (Reinert et al. in press).

Fluridone

Characteristics and use. Fluridone is a relatively new herbicide with herbicidal properties first reported in 1976 (Waldrop and Taylor 1976). Fluridone is being evaluated for aquatic weed control by Eli Lilly, Inc., under an Experimental Use Permit with the Environmental Protection Agency. Fluridone is effective for control of *Hydrilla* when applied to the water or to the

sediment surface. *Hydrilla* will absorb fluridone from the water through the leaves and shoots and from the sediments by way of the roots. Best results are achieved when *Hydrilla* is growing. Symptoms of injury may appear as soon as 7 days after treatment. One to two months may be required before full treatment effectiveness is evident. Reduction in dissolved oxygen concentration is not generally a problem with the slow action of this herbicide. Water chemistry and water quality are not appreciably altered in treated aquatic systems.

Fluridone apparently inhibits carotenoid synthesis. Carotenoids are both protective and accessory pigments; when carotenoid synthesis is inhibited, chlorophyll may be photodegraded and gradually destroyed. The plant appears chlorotic, especially the apical regions. *Hydrilla* loses its capacity for photosynthesis and eventually dies and decays.

Fluridone may be applied as a spray to the water's surface or to the sediment region. Fluridone may also be applied as pellets. Water flow should be stopped or minimized for two to three days for maximum effectiveness because of requisite contact time. Fluridone is a wide-spectrum herbicide and vegetation in close proximity to the treated area may be harmed if proper care is not exercised. Application rates of fluridone for *Hydrilla* control vary with water depth.

Sonar AS and Sonar 5P are the liquid and granular (respectively) fluridone formulations manufactured by Elanco Products Company (Table 1). Sonar AS is 45.2-percent fluridone and Sonar 5P is 5-percent fluridone (Table 8). Sonar is an experimental-use herbicide and is used in cooperation and in accordance with the experimental use permits. Sonar should be used in quiescent waters in order to ensure sufficient contact time to obtain control of *Hydrilla*. *Hydrilla* should be controlled when exposed to 0.5 mg/ for approximately 48 hr (Elanco 1981). Since fluridone is relatively soluble, it is doubtful that the Sonar 5P formulation would be a very slow release granule. Rather, Sonar 5P should probably be considered an approach to facilitate application of fluridone to an aquatic system when spray or injection equipment is not available.

Compartmentalization and persistence. Hydrolysis and volatilization of fluridone are relatively insignificant in aquatic systems (Table 9). In the pH range of most waters (4 to 11), fluridone is in the nonionized molecular form. The pH of the aquatic system should have little or no effect on

performance of fluridone. Solubility of fluridone is 12 mg/ℓ, and fluridone may sorb significantly to sediments with a reported Kp of 3.26 (Loh et al. 1979). With an octanol water partition coefficient of 74.1 and bioconcentration factors of 0.9 to 7.4 (West et al. 1983), relatively little fluridone should be found in fish or other aquatic organisms under normal application conditions. Biotransformation occurs, but environmentally relevant rate coefficients are not well known.

Photolysis is a major degradation pathway for fluridone in many aquatic systems. In systems with little dilution or dispersion, fluridone may persist for 4 to 55 days in the surficial sediment (West et al. 1983). A reported overall dissipation rate constant for fluridone in an aquatic system is 0.15/day (Muir et al. 1980). Because fluridone is a relatively new herbicide, there is little independent verification of these results. With wider use of fluridone, more information will become available as will precise and accurate rate coefficients.

## Herbicide Effects on *Hydrilla*

### Shoots and reproductive propagules

Diquat, endothall, various organic chelated forms of copper, and more recently fluridone are among the aquatic herbicides most widely used for management and control of *Hydrilla*. Also being used to control *Hydrilla* in fast-flowing irrigation canals in western states is acrolein (Anderson and Dechoretz 1982). However, this product is harmful to fish and wildlife at recommended application rates (Johnson 1970) and is not suggested for use in rivers and streams. Other registered aquatic herbicides that are effective on *Hydrilla* include dichlobenil and fenac (Steward 1980). Dichlobenil has a 90-day water-use restriction that severely limits its use. Fenac is registered for applications to exposed sediment following drawdown or draining. This restricts its use to areas that can be drained.

Under normal conditions, diquat, endothall, and copper act essentially as contact herbicides and kill green tissues rapidly; their translocation within the treated plants is limited. Consequently, these herbicides often destroy only the shoots, but the plants regrow quickly from rootcrowns and other propagules. Steward (1969) showed that 0.5 to 2.0 mg/ℓ of diquat or endothall caused no damage to *Hydrilla* propagules.

The impact of chemical control on nontarget submersed vegetation should be considered. Mackenzie and Hall (1967) reported that southern naiad (*Najas guadalupensis*) and coontail (*Ceratophyllum demersum*) were much more susceptible to diquat than *Hydrilla*. Similarly, Eurasian watermilfoil (*Myriophyllum spicatum*) was found to be susceptible to diquat at concentrations much lower than that required for *Hydrilla* control (Van, unpublished data*). Yeo (1970) reported that several pondweed species were susceptible to endothall at concentrations where control of *Elodea canadensis* could not be achieved.

*Hydrilla* decomposition and water quality

One of the impacts associated with the use of herbicides for the control of aquatic weeds is the effects of plant decomposition on water quality. The water-quality variables include the release of nitrogen and phosphorus, which might stimulate algal productivity within the vicinity of the treated area. The utilization of dissolved oxygen at night by this increased algal biomass, along with the decomposing aquatic weeds, could result in a lowering of dissolved oxygen concentrations. Based on the biomass values for *Hydrilla* in the Potomac River (Rybicki et al. 1985, Fig. 4) and the decomposition and nutrient release characteristics of native submersed species in the Chesapeak Bay (Twilley et al. in press), the lowering of dissolved oxygen concentration may be determined for the tidal Potomac River.

In order to evaluate this impact under a range of conditions, the average biomass of *Hydrilla* for the tidal Potomac River of 135 g (dry wt)/$m^2$ and a higher value of 500 g (dry wt)/$m^2$ were used to calculate nutrient release and oxygen consumption. Since the water depth in a plant bed is about 1 m, these biomass values are equivalent to 135 and 500 g (dry wt)/$m^3$. Nitrogen concentrations of aquatic plants in the Choptank River estuary (*Potamogeton perfoliatus* and *Myriophyllum spicatum*) were about 20 mg/g dry wt. Thus nitrogen standing crop is 2,700 to 10,000 mg/$m^3$. Total nitrogen release occurs for these macrophytes in about 14 days and is equivalent to only 7 percent of the total plant nitrogen content (Twilley et al. in press). This low release of nitrogen to the water column may be related to the limited supply of nitrogen in estuarine environments. Based on these values, total nitrogen release to the water column from decomposing *Hydrilla* would be about 14 to 50 μg/ℓ/day.

---

* Thai K. Van, USDA-Aquatic Plant Management Laboratory, Ft. Lauderdale, Fla., 1984.

Phosphorus release from decomposing aquatic macrophytes is generally higher than nitrogen release and may be related to the internal phosphorus content of the plant (Westerdahl 1981). Submersed macrophytes in the Choptank River estuary had phosphorus concentrations of 4 mg/g (dry wt) and 35 percent was released in 14 days during decomposition (Twilley et al. in press). Thus, phosphorus release to the water column from decomposing *Hydrilla* may be about 14 to 50 μg/ℓ/day, which is the same rate estimated for nitrogen release. For both nutrients, a major portion may remain in small particulate plant material (>80 percent for nitrogen and <50 percent for phosphorus) that accumulates and decomposes on the sediment surface. Sediments may also adsorb dissolved nutrients, especially phosphorus, from the water column. The turnover rate of these nutrients is still uncertain. The significance of these nutrient release rates is unclear because information on water depth and water velocity relationships is lacking.

Oxygen-consumption rates for decomposing *Potamogeton* and *Myriophyllum* under estuarine conditions ranged from 0.15 to 0.36 mg/g (dry wt)/hr. Based on an average of 0.26 mg/g (dry wt)/hr, decomposing *Hydrilla* may consume 0.96 to 3.12 mg/g (dry wt)/hr. The proper use of herbicides (i.e., recommended application rates, timing of application, and the total area of plants treated at a given time) will minimize any adverse impacts associated with nutrient release and depressed dissolved oxygen concentrations following plant decomposition.

Copper

Copper complexes are being used alone to control *Hydrilla* (Guppy 1967, Hearne and Pasco 1972). The effective concentrations, which are considerably higher than when applied in combination with diquat, are toxic to fish (Hearne and Pasco 1972). Sutton et al. (1972) showed that rates of copper could be reduced to 1 mg/ℓ when used with diquat. This combination has been used for *Hydrilla* control in Florida, California, and North and South Carolina.

Diquat

Early field tests in South Florida (Mackenzie and Hall 1967) indicated that diquat treatments at 0.5 mg/ℓ gave good control of *Hydrilla* in nonflowing water with low *Hydrilla* stands (less than 70-percent coverage). For the control of *Hydrilla* in dense infestations, however, higher treatment rates of 1 to 2 mg/ℓ diquat or combinations of diquat plus copper were required (Mackenzie and Hall 1967, Blackburn and Weldon 1970). Recently, with improved

techniques of herbicide placement (McClintock et al. 1974) and with the use of invert (Gates 1972) or polymer adjuvants (Wortley 1977), it is possible to control *Hydrilla* with 25 percent or less of the amount of diquat that would be required for the total water column treatment at 1.0 mg/ℓ (Bitting 1974, Baker et al. 1975).

Present recommendations for use of diquat to control *Hydrilla* include the addition of some forms of chelated copper. This combination has been shown by many authors to be more effective and safer than diquat or copper used alone (Mackenzie and Hall 1967, Blackburn and Weldon 1970). The increased phytotoxicity of this combination appeared to be related to increased uptake of both diquat and copper in the *Hydrilla* plant tissues (Sutton et al. 1970, 1972).

Endothall

Endothall is available for aquatic use as inorganic or amine salts. The amine formulation has been found to be most effective (Blackburn et al. 1971). Blackburn and Weldon (1970) showed that 2 to 4 mg/ℓ of this herbicide provided satisfactory control of *Hydrilla* in a series of laboratory and field evaluations. However, the long-chain amine salts are toxic to fish at concentrations of 0.3 to 1.0 mg/ℓ (Walker 1963). The dipotassium salt would seem, therefore, to be more desirable where fishery is the concern. Also, for weed control in a limited area or for spot or margin treatment, a granular formulation may be preferable. Sutton et al. (1971) reported a synergistic effect using a combination of 5.0 mg/ℓ endothall plus 1.0 mg/ℓ copper on *Hydrilla*.

Concentration vs. exposure time

Herbicides have been used successfully for the management of *Hydrilla* in static and slow-moving water where contact with the herbicide could be maintained for several days or weeks. The control of *Hydrilla* in flowing water, however, is far more difficult because the herbicide is rapidly washed away from the application site and the necessary contact time may not be achieved.

In general, there is very little information on the minimum contact time required for effective weed control. Mackenzie (1968) observed that control of *Hydrilla* was obtained with diquat at 0.5 to 1.0 mg/ℓ only where the water was static and where heavy rainfall did not dilute the treatment within 48 hr after application. Barrett (1981) stated that in Britain the recommended treatment rate of diquat for control of submersed vegetation is 1.0 mg/ℓ with a minimum contact time of 24 hr. Preliminary results from laboratory

experiments indicated that 24-hr contact time was also required to control *Hydrilla* with 1.0 mg/ℓ diquat (Van, unpublished data*).

Studies on three copper formulations showed that a chelated compound (Komeen ®) was effective as a contact herbicide on *Hydrilla* at 2.0 mg/ℓ for 4 to 6 hr or at 4.0 mg/ℓ for 2 to 4 hr (Anderson et al. 1984).

Label recommendations for the use of endothall to control *Hydrilla* in irrigation and drainage canals in Florida specify a minimum contact time of 2 hr at 3 to 5 mg a.e.(acid equivalent)/ℓ. Price (1969) applied the amine salt at 3 to 4 mg/ℓ for 3 hr in canals in western states and reported good control of several pondweed species for a distance of 30 km downstream. However, a similar treatment of 6 mg a.e./ℓ of endothall amine for 3 hr provided only limited control of *Elodea canadensis* in flowing water in the Berriquin Irrigation District in Australia (Bowmer et al. 1979). Also, using an exposure time of 3 hr in a static assay on *E. canadensis*, Bowmer and Smith (1984) reported that acrolein at 3 mg/ℓ gave 80-percent reduction in biomass, whereas for endothall this level of control was not reached even by concentrations exceeding 100 mg/ℓ. For the control of *Hydrilla*, laboratory experiments on herbicide concentration with time indicated that the minimum contact time could be decreased from 48 to 12 hr by increasing the dipotassium endothall treatment rate from 1.0 to 3.0 mg a.e./ℓ (Van, unpublished data*). To kill *Hydrilla* within an approximate 2- to 4-hr contact time as expected in the tidal Potomac River, the maximum label rate of 5 mg/ℓ endothall would be required. The 2- to 4-hr contact time was assumed to be the period of time around low tide during which the flow velocity is minimal (see Chapter 1).

Uptake characteristics

The success of high concentration/short exposure-time treatments in flowing water depends on the relatively rapid uptake and retention of a lethal quantity of herbicide by the plant. However, information on herbicide uptake and lethal concentration in plant tissues is extremely limited for aquatic macrophytes, especially in submersed species.

Generally, the slow-acting translocated herbicides appear to have much slower uptake rates. For example, a minimum herbicide concentration must be maintained in the water for several days and sometimes weeks to ensure the

---

* Thai K. Van, USDA-Aquatic Plant Management Laboratory, Ft. Lauderdale, Fla., 1984.

effectiveness of fluridone in controlling pondweeds (Anderson 1981) and *Hydrilla* (Hall et al. 1984). Contact herbicides, on the other hand, are taken up rapidly and therefore appear more suitable for use in flowing water. Davies and Seaman (1968) reported that uptake of diquat by *E. canadensis* consists of an initial rapid adsorption phase followed by a constant active uptake phase that continued over a 4.5-hr experiment. Sutton et al. (1972) observed a linear uptake of diquat in *Hydrilla* shoots that continued for 9 days. Thomas and Seaman (1968) using $^{14}C$-labeled endothall observed uptake of the herbicide by both the foliage and root tissues in American pondweed (*Potamogeton nodosus*). These authors also recorded movement of the $^{14}C$ label from mature photosynthesizing leaves and accumulation of the herbicide in the apices and developing secondary shoots. However, there was no movement of the $^{14}C$ label from the treated roots to the foliage of these plants, possibly because of the lack of the transpiration stream in submersed aquatic plants. Haller and Sutton (1973) found endothall to accumulate in the apices of *Hydrilla* faster than in lower portions of the stem. The addition of copper sulfate at 0.4 to 2.0 μM increased endothall absorption but higher concentrations of 4.0 to 16.0 μM inhibited endothall uptake. Uptake was inhibited at lower temperatures (10° C) relative to higher temperatures (20° and 30° C). More endothall was absorbed in the light than in darkness.

One major problem with most of the herbicide uptake studies was the lack of information on the required lethal concentration in plant tissues and the minimum exposure time required to attain that concentration for effective weed control. This information is essential for the development of a herbicide management program to control *Hydrilla* in flowing water.

## Herbicide Effects on Nontarget Organisms

The following discussion concerns primarily the acute and chronic toxicity of copper, diquat, endothall, and fluridone on nontarget organisms. Acute toxicity tests involve short-term exposure of the organism to different concentrations of the chemical. The most commonly measured effect is lethality (death). Chronic toxicity tests are concerned with evaluating lethal or sublethal effects resulting from long-term exposure to low concentrations of a specific chemical. The sublethal effects may include physiological, biochemical, behavioral, and histological changes; mutagenicity;

carcinogenicity; or teratogenicity. Generally, chronic studies are conducted if (1) the mammalian or avian data suggest the chemical may influence reproduction; (2) the chemical has a high potential for bioaccumulation and persistence; (3) the acute toxicity ($LC_{50}$) is less than 1.0 mg/ℓ; and (4) the estimated environmental concentration of the chemical is greater than 1/100 the $LC_{50}$ for specific nontarget organisms.

Acute and chronic toxicity of copper

Acute toxicity tests for copper have been conducted on 18 invertebrate and 27 fish species. Most of the fish toxicity tests have been conducted with four salmonid species, fathead minnows, and bluegills.

Freshwater species. The acute toxicity values for freshwater organisms range from a low of 7.24 μg/ℓ for *Daphnia pulicaria* in soft water to 10,200 μg/ℓ for bluegills in hard water (Table 10, reproduced from US Environmental Protection Agency (USEPA) 1980). The toxicity of copper decreases with an increase in water hardness, alkalinity, and total organic carbon. The range of acute values indicates that some of the more resistant species could survive at copper concentrations 100 times greater than those that would be lethal to the more sensitive species. Among the more sensitive species are daphnids, scuds, midges, and snails, which form the major food chains for both warm- and cold-water fishes. The concentrations of copper acutely lethal to these sensitive organisms in soft water are only slightly above those chronically toxic to most fish and invertebrate species.

The data on the chronic toxicity of copper to freshwater organisms are available for 15 freshwater species (4 invertebrates and 11 fish species) (Table 11, reproduced from USEPA 1980). The chronic toxicity values range from a low of 3.9 μg/ℓ for early life stage tests with brook trout in soft water to 60.4 μg/ℓ for a similar test with northern pike in hard water. Fish and invertebrate species seem to be about equally sensitive to the chronic toxicity of copper. Hardness does not appear to affect the chronic toxicity of copper.

Saltwater species. The acute toxicity of copper to saltwater species ranges from 17 μg/ℓ for the calenoid copepod to 600 μg/ℓ for the shore crab. The saltwater invertebrate data include investigations with three phyla: annelids, moluscs, and arthropods (crustaceans). The acute values for saltwater fish include data for four species. Acute toxicity ranged from 28 μg/ℓ for summer flounder embryos to 510 μg/ℓ for the Florida pompano. In a chronic

life-cycle test with mysid shrimp, adverse effects were noted at 77 μg/ℓ but not at 38 μg/ℓ.

Mutagenicity, carcinogenicity, and teratogenicity. Copper is not known to have mutagenic, carcinogenic, or teratogenic properties.

EPA water-quality criteria. For total recoverable copper, the criterion to protect freshwater aquatic life according to the EPA guidelines is 5.6 μg/ℓ as a 24-hr average and the concentration (in μg/ℓ) should not exceed the numerical value given by (0.94 [ℓn(hardness)] - 1.23) at any time. For example, at a hardness of 50, 100, and 200 mg/ℓ as $CaCO_3$, the concentration of total recoverable copper should not exceed 12, 22, and 43 μg/ℓ, respectively, at any time. For total recoverable copper, the criterion derived according to the EPA guidelines to protect saltwater organisms is 4.0 μg/ℓ as a 24-hr average; the concentration should not exceed 23 μg/ℓ at any time.

Acute and chronic toxicity of endothall

Freshwater species. The data on the acute toxicity of endothall (dipotassium or disodium salt), as described herein, have been summarized previously (Pennwalt Corp. 1984) and appropriate tables (Tables 12-16), reproduced and included herein. The acute toxicity ranged from 82 mg/ℓ in Chinook salmon to 450 mg/ℓ in rainbow trout (Table 12). These concentrations are substantially higher than those expected to be found in the field. Toxicity studies using cancer exposures showed no increase in toxicity (Table 13). Any toxicity to fish and invertebrates may result indirectly from oxygen depletion due to decaying vegetation if the herbicide is not applied correctly.

In contrast to the dipotassium salt of endothall, the N,N'-dimethylalkylamine salt of endothall was more toxic to fish; the 96-hr $LC_{50}$ for several freshwater fish is less than 1 mg/ℓ (0.14-0.98 mg/ℓ) (Table 14). The N,N'-dimethylalkylamine salt appears to be highly toxic to aquatic invertebrates; the 96-hr $LC_{50}$ values of this herbicide formulation for two species of amphipod, *Gammarus fasciatus* and *G. lacustris*, and the grass shrimp, *Palaemonete* sp., were 0.51, 0.50, and 0.05 mg/ℓ, respectively (Johnson and Finley 1980). Though safer to use, the margin of safety is less than with the dipotassium salt of endothall.

Several $LC_{50}$ values have been reported for freshwater aquatic invertebrates (Table 15). As observed with $LC_{50}$ values for fish, mortality is produced only at values far in excess of labelled application rates. Field

observations and laboratory studies have demonstrated no adverse impact on treatment-area fauna (Table 16).

Saltwater species. Acute toxicity values (Table 15) for dipotassium endothall are available for only two species, the eggs and larvae of hard clams and oysters. The data on the toxicity of the herbicide to the adults of the two species are not available.

The information available on the subchronic or chronic toxicity of endothall to freshwater or saltwater organisms is extremely limited. In a 0.31-ha pond treated with 5.0 mg/ℓ of dipotassium endothall, the herbicide did not affect the number of young-of-the-year bluegills produced by the original adult stock during the year of the treatment and the year following treatment and did not affect the reproduction of first-generation bluegills. The survival of adult and first-generation bluegills was not affected (Serns 1977).

The above data indicate that dipotassium endothall has a sufficient safety margin for the aquatic organisms tested.

Mutagenesis, carcinogenesis, and teratogenesis. There are no data in the literature to suggest that endothall is mutagenic, carcinogenic, or teratogenic.

Acute and chronic toxicity of diquat

Freshwater species. Information on the acute toxicity of diquat is available for nine species of freshwater fish. The toxicity of the herbicide to these fish ranges from 2.1 mg/ℓ in walleye to 245 mg/ℓ in bluegills (Table 17).

Like fish, freshwater amphipods show considerable variation in their sensitivity to diquat. *Hyelella* is extremely sensitive (96-hr mean total lethality (TLm), 0.048 mg/ℓ) to the herbicide. On the other hand, the *Gammarus* (amphipods) and mayfly larvae were quite resistant to the herbicide with a 96-hr $LC_{50}$ of 16.4 and >100 mg/ℓ, respectively.

Saltwater species. Information on the acute toxicity of diquat to saltwater organisms is available for only two species, shrimp and cockle. In each case, the 48-hr $LC_{50}$ was greater than 10 mg/ℓ. The herbicide at a concentration on 1 ppm had no effect on white shrimp, oysters, and longnose shrimp following a 48-hr exposure.

No information is available on the subchronic or chronic toxicity of diquat (fish early-life stage, aquatic invertebrate life cycle, and fish life cycle) to freshwater or saltwater organisms under controlled laboratory

conditions. However, in a chronic study conducted in pools stocked with fingerlings and adult bluegills, applications of 1 or 3 mg/ℓ diquat at intervals did not affect the survival of either group of fish (Gilderhaus 1967).

Since a significantly high proportion of diquat applied to water tends to associate with the sediment, particular consideration should be given to assessing the effects of the herbicide on benthic organisms.

Mutagenicity, carcinogenicity, and teratogenicity. There is no evidence in the literature to suggest that diquat has mutagenic, carcinogenic, or teratogenic effects.

Acute and chronic toxicity of fluridone

Freshwater species. The 96-hr $LC_{50}$ for four species of freshwater fish (trout, bluegills, catfish, and fathead minnows) ranges from 7.6 to 22 mg/ℓ. The concentration is approximately 76 to 220 times the normal application rate. Some invertebrates are more sensitive than fish. The 48-hr $EC_{50}$ values for *Daphnia* and midge larvae are 3.4 and 1.3 mg/ℓ, respectively.

Data on the chronic toxicity are available for several species. Several months of continuous exposure of catfish eggs and the resulting larvae to a constant concentration of 0.5 mg/ℓ produced no adverse effect. No adverse effects were noticed during the full life-cycle (egg-to-egg) test with fathead minnow at a concentration of 0.48 mg/ℓ. A concentration of 0.6 mg/ℓ had no effect on the growth or survival of amphipods or on the emergence of the adult midges. Reproduction of *Daphnia* was not affected by a concentration of 0.2 mg/ℓ fluridone.

Saltwater species. No information is available on the acute, subchronic, or chronic toxicity of fluridone to saltwater organisms.

Mutagenicity, carcinogenicity, and teratogenicity. On the basis of the available data, fluridone does not induce mutagenic, carcinogenic, or teratogenic effects.

## Conclusions and Recommendations

Herbicides effective on *Hydrilla* were considered in arriving at preliminary recommendations. These herbicides are listed and reasons for acceptance or rejection are summarized in Tables 18 and 19.

The herbicides were grouped into three categories based on the criteria noted. The first category contained herbicides that are presently not acceptable for use in the Potomac River. This category included arcolein, dichlobenil, and fenac.

Other candidate herbicides were grouped in a second category because they may lack adequate toxicological testing. This category contained copper complexes, endothall, fluridone, and a tank mix of copper and diquat.

If the water in the areas infested with *Hydrilla* is soft, the toxicity of copper to vertebrates and invertebrates would probably be unacceptable. Copper is not subject to biodegradation and, once introduced to an aquatic ecosystem, it remains within the sediment, water, or biota until it is physically transported from the system.

Based on toxicological and fate considerations, endothall was initially considered a prime candidate for use in the Potomac River. However, endothall was eliminated as a potential herbicide since it is not registered for use in flowing water. Additional testing is being conducted by Pennwalt Corporation; however, it will be approximately 4 yr before the tests will be completed.

Fluridone is an experimental herbicide and unavailable for use in the Potomac River. In addition, the use of fluridone in flowing water may not be effective since the contact time with the plant may be insufficient to obtain adequate control.

For operational control of *Hydrilla*, diquat is the only herbicide available for use in flowing water and was placed in a third category. Diquat is effective on *Hydrilla* and is registered for use in flowing water. However, there remain a number of questions regarding effectiveness of this herbicide in the Potomac River, and those questions should be resolved prior to initiating a large-scale treatment program. With this information and the in-situ suspended solids and flow velocity data, a decision can be made on recommending the use of diquat in the Potomac River.

a. The first question concerns the organic/inorganic content of the suspended particulate within designated treatment areas. The composition of

the suspended particulate must be determined before the effectiveness of diquat can be determined for this environment. Specifically, the level of inorganic matter in the suspended particulate should be determined during the *Hydrilla* growing season and the period in which treatment would be most likely (i.e., late May through early September). If the inorganic portion of the suspended particulates is high, then diquat probably should not be used due to adsorption of diquat to the clay particles.

b. Another important question concerns the flow velocity within the treatment areas. Herbicide concentration/exposure time is dependent on flow velocity. If the flow velocity is high 2 hr prior to and after low tide slack water, then diquat would not be considered a good choice. Concentrations of diquat required to control *Hydrilla* will also kill numerous other species within the immediate treated area.

c. A major problem with using diquat in the Potomac River is the potential for rapid dispersal of the herbicide out of the treated area. Consequently, the following information is needed:

- Minimum diquat concentration/exposure time required to control *Hydrilla*.
- Rate of diquat uptake by *Hydrilla*.
- Flow velocity range within which diquat can be shown to be efficacious toward *Hydrilla*.

d. At the request of the US Army Engineer District, Baltimore, the Aquatic Plant Control Operations Support Center of the US Army Engineer District, Jacksonville, compiled information of various control techniques used to manage dioecious *Hydrilla*. The information was based on Jacksonville District experiences and a limited literature survey. A tabulation of the information is contained in Appendix B.

## Contributors


| | |
|---|---|
| Dr. David C. Beckett | US Army Engineer Waterways Experiment Station<br>ATTN: WESER-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. Kurt D. Getsinger | US Army Engineer Waterways Experiment Station<br>ATTN: WESES-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. Donald Graetz | Soil Science Dept.<br>106 Newell Hall<br>University of Florida<br>Gainesville, FL 32611 |
| Mr. Jerry F. Hall | US Army Engineer Waterways Experiment Station<br>ATTN: WESES-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |
| Dr. John Rodgers, Jr. | Institute of Applied Sciences<br>North Texas State University<br>Denton, TX |
| Dr. Harish Sikka | Director, Great Lakes Research Center<br>Suny Lab<br>1300 Allenwood Ave<br>Buffalo, NY 14222 |
| Dr. Robert Twilley | Horn Point Laboratories<br>PO Box 775<br>Cambridge, MD 21613 |
| Dr. Thai K. Van | USDA/ARS<br>3205 SW College Avenue<br>Ft. Lauderdale, FL 33314 |
| Dr. Howard E. Westerdahl* | US Army Engineer Waterways Experiment Station<br>ATTN: WESES-A<br>PO Box 631<br>Vicksburg, MS 39180-0631 |


---

* Dr. Westerdahl was primarily reponsible for the development of this chapter and should be used as a point of contact regarding its content.

## References


Applied Biochemists, Inc. 1983. 5300 West County Line Rd., 96 North, Mequon, Wisconsin 53092.

Anderson, L. W. J. 1981. Effect of light on the phytotoxicity of fluridone in American pondweed (*Potamogeton nodosus*) and sago pondweed (*P. pectinatus*). Weed Sci. 29:723-728.

Anderson, L. W. J., and N. Dechoretz. 1982. Growth, reproduction, and control of *Hydrilla verticillata* Royle (L.F.) in an irrigation system in South Western U. S. Proc. EWRS 6th. Inter. Symp. on Aquatic Weeds, Novi Sad, Yugoslavia, pp 54-61.

Anderson, L. W. J., N. Dechoretz, G. Darmstadt, and D. L. Bayer. 1984. Uptake and efficacy of three copper formulations on *Hydrilla verticillata* (L.F.) Royle. Weed Sci. Soc. Amer. Abstract.

Ashton, F. M., and A. D. Crafts. 1981. Mode of Action of Herbicides. Wiley-Interscience, New York, 504 pp.

Audus, L. J. (ed.) 1976. Herbicide Physiology, Biochemistry and Ecology. 2nd ed. Academic Press, New York. Vols. 1 and 2. 308 and 364 pp.

Baker, G., L. E. Bitting, P. A. Lambert, W. L. McClintock, and W. D. Hogan. 1975. Improved application techniques for aquatic herbicides. Hyacinth Control J. 13:21-24.

Barrett, P. R. F. 1981. Aquatic herbicides in Great Britain, recent changes and possible future development. Proc. Aquatic Weeds and Their Control, Association of Applied Biologists, Oxford. pp 95-103.

Bitting, L. E. 1974. Field uses of invert emulsions. Hyacinth Control J. 12:34-35.

Blackburn, R. D., and L. W. Weldon. 1964. Control of southern naiad in Florida drainage and irrigation channels. Weeds 12:295-298.

Blackburn, R. D., and L. W. Weldon. 1970. Control of *Hydrilla verticillata*. Hyacinth Control J. 8:49.

Blackburn, R. D., K. A. Boyer, and C. E. Timmer. 1971. Phytotoxicity of four formulations of the alkylamine salt of endothall on *Hydrilla verticillata* and fish. Hyacinth Control J. 9:55-58.

Blackburn, R. D., and L. W. Weldon. 1970. Control of *Hydrilla verticillata*. Hyacinth Control J. 8:4-9.

Bowmer, K. H., G. R. Sainty, G. Smith, and K. Shaw. 1979. Management of elodea in Australian irrigation systems. J. Aquat. Plant Manage. 17:4-12.

Bowmer, K. H., and G. H. Smith. 1984. Herbicides for injection into flowing water: acrolein and endothal-amine. Weed Res. 24:201-211.

Brooker, M. P., and R. W. Edwards. 1973. Effects of the herbicide paraquat on the ecology of a reservoir. Freshwater Bio. 3:157-175.

Butler, P. A. 1965. Commercial fisheries investigation. In: Pesticides - Wildlife Studies, A review of Fish & Wildlife investigations during 1961 and 1962. USF&WS Circ. 167 pp 26-42.

Calderbrank, A. 1968. The bipyridylium herbicides. Adv. in Pest Control Res. 8:127-235.

Cedeno-Maldonado, A., and H. A. Swader. 1974. Studies on the mechanism of copper toxicity to *Chlorella*. Weed Sci. 22:443-449.

Coats, G. E., H. H. Fanderburk, Jr., J. M. Lawrence, and D. E. Davis. 1966. Factors affecting persistence and inactivation of diquat and paraquat. Weed Res. 6:58-66.

Coats, G. E., H. H. Funderburk, Jr., J. M. Lawrence, and D. E. Davis. 1964. Persistence of diquat and paraquat in pools and ponds. Proc. South. Weed Conf. 17:308-320.

Crafts, A. S. 1975. Modern Weed Control. University of California Press, Berkeley, 440 pp.

Folmar, L. C. 1977. Technical paper No. 88, Fish and Wildlife Service, U. S. Department of the Interior, Washington, DC.

Frank, P. A., and R. D. Comes. 1967. Herbicidal residues in pond water and hydrosoil. Weeds 16:210-213.

Gates, R. J. 1972. Control of hydrilla using the bifluid system. Hyacinth Control J. 10:28-29.

Gilderhaus, P. A. 1967. Effects of diquat on bluegills and their food organisms. Prog. Fish. Cult. 29:67-74.

Grzenda, A. R., H. P. Nicholson, and W. S. Cox. 1966. Persistence of four herbicides in pond water. J. Am. Water Works Assoc. 58:326-332.

Guppy, R. H. 1967. A comparison of various methods for the control of Florida elodea in Orange County, Florida. Hyacinth Control J. 6:24-25.

Hall, J. F., H. E. Westerdahl, and T. J. Stewart. 1984. Growth response of *Myriophyllum spicatum* and *Hydrilla verticillata* when exposed to continuous, low concentration of fluridone. Technical Report A-84-1, U. S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Miss., 31 pp.

Haller, W. T., and D. L. Sutton. 1973. Factors affecting the uptake of endothall-$^{14}C$ by hydrilla. Weed Sci. 21:446-448.

Haven, D. S. 1969. Levels of the herbicide diquat in two estuarine molluscs and in the water and mud. The Va. J. Sci. 20:51-53.

Hearne, J. S., and R. A. Pasco. 1972. Aquatic weed control trials, Gatun Lake, Panama Canal. Hyacinth Control J. 10:33-35.

Hiltibran, R. C. 1963. Effect of endothall on aquatic plants. Weeds 11:256-257.

Hiltibran, R. C. 1967. Effects of Some Herbicides on Fertilized Fish Eggs and Fry. Trans. Amer. Fish. Soc., Vol 96, pp 414-416.

Hiltibran, R. C., D. L. Underwood, and J. S. Fickle. 1972. Fate of diquat in the aquatic environment. Univ. of Illinois Water Resources Center, WRC Report No. 52. 50 pp.

Holmberg, D. J. 1973. The effects and persistence of endothall in the aquatic environment. Madison, Wisc.: University of Wisconsin. 127 pp. Thesis.

Holmberg, D. J., and G. F. Lee. 1976. Effects and persistence of endothall in the aquatic environment. J. Water Poll. Contr. Fed. 48:2738-2746.

Johnson, C. P. 1970. The commercial application of acrolein. Hyacinth Control J. 8:53.

Johnson, W. W., and M. T. Finley. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. Resource Publication 137. US Department of Interior, Fish and Wildlife Service, Washington, DC.

Jones, R. O. 1965. Tolerance of the fry of common warm-water fishes to some chemicals employed in fish culture. Proceedings, 16th Annual Conference Southeast Assoc. Game Fish Commission. 1962. pp 436-445.

Kearney, P. C., and D. D. Kaufman (eds.). 1976. Herbicides - Chemistry, Degradation and Mode of Action. 2nd ed. New York: Marcell Dekker, Inc. Vols. 1 & 2, 500 and 536 pp.

Khan, S. U. 1974. Adsorption of bipyridylium herbicides by humic acid. J. Environ. Quality 3:202-206.

Kilgore, K. J. 1981. Eurasian watermilfoil treatment with 2,4-D, diquat and endothall at reduced application rates. In: Proc. 15th Annual Mtg. Aquatic Plant Control Research and Ops Rev. Miscellaneous Paper A-81-3, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss. pp 450-463.

Klingman, G. C., F. M. Ashton, and L. J. Noodhoff. 1975. Weed Science: Principles and Practices. New York: Wiley. 431 pp.

Lawrence, J. M., et al. 1962. Aquatic herbicide data. U. S. Department of Agriculture, Agriculture Handbook No. 131. 133 pp.

Loh, A., S. J. Parka, R. Albritton, and C. C. Lin. 1979. Use of adsorption coefficients and soil properties to predict fluridone herbicidal activity. Weed Sci. 27:456-459.

MacKenzie, J. W. 1968. Factors affecting control of Florida elodea with diquat in southeast Florida canals. Hyacinth Control J. 8:13-16.

MacKenzie, J. W., and L. H. Hall. 1967. *Hydrilla* control in southeast Florida with diquat. Hyacinth Control J. 6:37-44.

Maestri, M. 1967. Structural and functional effects of endothall on plants. Davis, California: University of California. 100 pp. Dissertation.

McClintock, W. L., J. W. Frye, and W. D. Hogan. 1974. Development of bottom placement technique for *Hydrilla* and eelgrass control. Hyacinth Control J. 12:46-47.

McCowan, M. C., C. L. Young, S. D. West, S. J. Parka, and W. R. Arnold. 1979. Fluridone, a new herbicide for aquatic plant management. J. Aquat. Plant Manage. 17:27-30.

Muir, D. C. G., N. P. Grift, A. P. Blouw, and W. L. Lockhart. 1980. Persistence of fluridone in small ponds. J. Environ. Qual. 9:151-156.

Narine, D. R., and R. D. Grey. 1982. Binding of diquat and paraquat to humic acid in aquatic environments. Soil Sci. 133:356-363.

Neely, W. B., and D. Mackay. 1981. An evaluative model for estimating environmental fate. In: Dickson, Maki, and Cairns (eds.). Modeling the Fate of Chemicals in the Aquatic Environment, Ann Arbor Science, Ann Arbor, Mich. 127 pp.

Pennwalt Corporation. 1984. Review of the effects of endothall products on aquatic ecosystems. Miscellaneous Report, Published by Pennwalt Corporation, 3 Parkway, Philadelphia, Pa. 19102.

Poinke, H. B., and G. Chesters. 1973. Pesticide-sediment-water interactions. J. Environ. Quality 2:29-45.

Price, A. 1969. The use of an amine salt of endothall in irrigation canals. Hyacinth Control J. 8:32-33.

Reinert, K. H., and J. H. Rodgers, Jr. 1984. Influence of sediment types on the sorption of endothall. Bull. Environ. Contam. Toxicol. 32:557-564.

Reinert, K. H., S. Stewart, M. L. Hinman, J. H. Rodgers, Jr., and T. J. Leslie. In Press. Release of endothall from aquathol granular aquatic herbicide. Water Research.

Rodgers, J. H., Jr., K. H. Reinert, and M. L. Hinman. 1984. Water quality monitoring in conjunction with the Pat Mayse Lake Aquatic Plant Management Program. In: Proceedings, 18th Annual Meeting, Aquatic Plant Control Research Program. Miscellaneous Paper A-84-4, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss. pp 17-24.

Rybicki, N. B., V. Carter, R. T. Anderson, and T. J. Trombley. 1985. *Hydrilla verticillata* in the tidal Potomac River, 1983 and 1984. USGS Open-file Report (in press).

Serns, S. L. 1977. Effects of dipotassium endothall on rooted aquatics and adult and first generation blue gills. Water Resources Bulletin 13:71-80.

Simsiman, G. V., and G. Chesters. 1976. Persistence of diquat in the aquatic environment. Res. Rev. 62:131-174.

Steward, K. K. 1969. Effects of growth regulators and herbicides on germination of *Hydrilla* turions. Weed Sci. 17:299-301.

Steward, K. K. 1980. Retardation of hydrilla (*Hydrilla verticillata*) regrowth through chemical control of vegetative reproduction. Weed Sci. 28:245-251.

Strange, R. J., and C. B. Shreck. 1976. Response of aerobic community metabolism to chemical treatment of aquatic macrophytes. J. Aquat. Plant Manage. 4:45-55.

Summers, L. A. 1980. The Bipyridinium Herbicides. Academic Press, New York. 449 pp.

Surber, E. W., and Q. H. Pickering. 1962. Acute toxicity of endothall, diquat, hyamine, delapon and silvex to fish. Progressive Fish Culturist 24:164-171.

Sutton, D. L., W. T. Haller, K. K. Steward, and R. D. Blackburn. 1972. Effect of copper on uptake of diquat-$^{14}C$ by *Hydrilla*. Weed Sci. 20:581-583.

Sutton, D. L., L. W. Weldon, and R. D. Blackburn. 1970. Effect of diquat on uptake of copper in aquatic plants. Weed Sci. 18:703-707.

Thomas, T. M. 1966. Uptake and fate of endothall in submerged aquatic plants. Davis, California: University of Calif. 94 pp. Thesis.

Thomas, T. M., and D. E. Seaman. 1968. Translocation studies with endothal-$^{14}C$ in *Potamogeton nodosus* Poir. Weed Res. 8:321-326.

Tucker, B. V., B. V. Peck, and J. N. Openson. 1967. Absorption of bipyridylium herbicides in soil. J. App. Food Chem. 15:1005-1008.

Twilley, R. R., W. M. Kemp, K. W. Staver, J. C. Stevenson, and W. R. Boynton. In Press. Nutrient enrichment of estuarine submersed vascular plant communities. I. Algal growth and effects on production of plants and associated communities. Marine Ecology Progress Series.

US Environmental Protection Agency. 1980. Ambient water quality criteria for copper. Report No. 440/5-80-36, Washington, DC.

Waldrop, T. W., and H. M. Taylor. 1976. 1-Methyl-3-phenyl-5-[3-trifluoromethyl)-phenyl]-4(1H)-pyridinone, a new herbicide. J. Agric. Food Chem. 24:1250-1251.

Walker, C. R. 1963. Endothall derivatives as aquatic herbicides in fishery habitats. Weeds 11:226-232.

Weed Science Society of America. 1970. Herbicide Handbook. 2nd ed. WSSA. New York, N.Y.

Weed Science Society of America. 1983. Herbicide Handbook. 5th ed. WSSA. Champaign, Ill. 513 pp.

Welborn, T. L. 1969. The toxicity of nine therapeutic and herbicidal compounds to striped bass. Progressive Fish Culturist 31(1):27.

West, D. S., R. O. Burger, G. M. Poole, and D. H. Mowrey. 1983. Bioconcentration and field dissipation of the aquatic herbicide fluridone and its degradation products in aquatic environments. J. Agric. Food Chem. 31:579-585.

Westerdahl, H. E. 1981. Aquatic plant control in the Panama Canal: field evaluation of endothall in Gatun Lake, Panama. Proc. 15th Annual Mtg. Aquatic Plant Control Research. Miscellaneous Paper A-81-3, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Wilson, D. C., and C. E. Bond. 1969. The effects of the herbicides diquat and dichlobenil (Casoron) on pond invertebrates. Trans. Am. Fish. Soc. 98:438-443.

Wortley, G. E. 1977. The use of polymers in the application of aquatic herbicides. Proc. South Weed Sci. Society 30:345-351.

Yeo, R. R. 1967. Dissipation of diquat and paraquat, effects on aquatic weeds and fish. Weeds 15:42-46.

Yeo, R. R. 1970. Dissipation of endothall and effects on aquatic weeds and fish. Weed Sci. 18:282-284.

Zepp, R. G., N. L. Wolfe, J. A. Gordon, and G. L. Baughman. 1975. Dynamics of 2,4-D esters in surface waters. Env. Sci. Technol. 9:1144-1149.

Table 1

List of Recommended Registered Aquatic Herbicides and Manufacturers

| Type | | Source |
|---|---|---|
| Copper ion, complexes | Cutrine-Plus (Liquid, 9% Cu, ethanolamine complex) | Applied Biochemists, Inc.<br>P. O. Box 25<br>Mequon, WI 53092 |
| | K-TEA Algaecide (Liquid, 8% Cu, triethanolamine complex) | Kocide Chemical Corp.<br>P. O. Box 45539<br>12701 Almeda Road<br>Houston, TX 77045 |
| | Komeen/Koplex (Liquid, 8% Cu, ethylenediamine complex) | Kocide Chemical Corp.<br>P. O. Box 45539<br>12701 almeda Road<br>Houston, TX 77045 |
| Diquat | Diquat Water Weed Killer (Liquid, 35.3%) | Ortho Division<br>Chevron Chemical Co.<br>940 Hensley St.<br>Richmond, CA 94804 |
| Endothall | Aquathol (Granular, 10.1% $K_2$ salt) | Pennwalt Corporation<br>Agchem Division<br>P. O. Box 6000<br>Concordville, PA 19331 |
| | Aquathol K (Liquid, 40.3% $K_2$ salt) | Pennwalt Corporation<br>Agchem Division<br>P. O. Box 6000<br>Concordville, PA 19331 |
| Fluridone | Sonar AS (Liquid, 45.2%) | Elanco Products Co.<br>740 S. Alabama St.<br>Indianopolis, IN 46285 |
| | Sonar 5P (Granular, 5%) | Elanco Products Co.<br>740 S. Alabama St.<br>Indianopolis, IN 46285 |

Table 2

Copper Ion Complexes - Herbicide Information

Common and Trade Names:

Copper

CUTRINE-PLUS (liquid, 9% Cu, ethanolamine complex)

K-TEA ALGAECIDE (liquid, 8% Cu, triethanolamine complex)

KOPLEX AQUATIC HERBICIDE (liquid, 8% Cu, ethylenediamine complex)

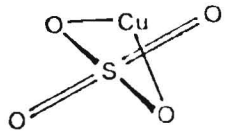
KOMEEN (liquid, 8% Cu, ethylenediamine complex)

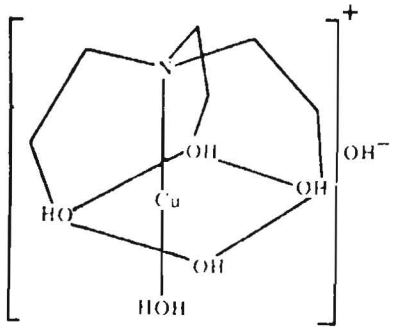
I.U.P.A.C. Name:

Copper Ion Complex

Formula and Structure:

$CuSO_4$, $Cu^{++}$




Mode of Action:

1. Cell toxicant.
2. $Cu^{++}$ inhibits electron transport system in photosystems I and II (Cedeno-Maldonado and Swader 1974).
3. Binds cytochrome C in electron transport system.

Table 3

Environmental Rate Constants for Copper Complexes

| Process | Rate |
|---|---|
| $K_T$ (overall) | Infinite persistence |
| $t_{1/2}$ | |
| Photolysis | Stable |
| Oxidation | -- |
| Hydrolysis | -- |
| Volatilization | Not volatile |
| Sediment sorption | Important process for organic sediments, precipitates on clays, forms insoluble copper hydroxides, phosphates, or carbonates. |
| Water solubility | Soluble |
| Bioconcentration | -- |
| Biotransformation and biodegradation | No metabolism |

Table 4

Diquat Dibromide - Herbicide Information

Common and Trade Names:

Diquat

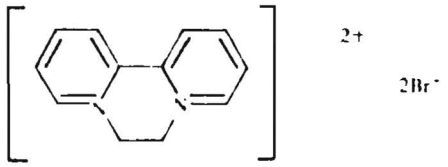
DIQUAT WATER WEED KILLER (liquid, 35.3%)

I.U.P.A.C. Name:

6,7-Dihydrodipyrido (1,2-a:2'-1'-C) pyrazinedinium ion

Formula and Structure:

$C_{12}H_{12}N_2X_2$ where X = Br




Mode of Action:

1. Acts as electron acceptor during the Hill reaction of photosynthesis; forms free radicals (Calderbrank 1968)
2. Free radicals oxidize and form hydrogen peroxide, which accumulates and destroys plant cells (Weed Society of America (WSSA) 1983)

Table 5

Environmental Fate Rate Constants for Diquat

| Process | Rate |
|---|---|
| $K_T$ (overall) overall disappearance rate coefficient | 0.75/day (calculated from Grzenda et al. 1966)<br>0.925/day (calculated from Frank and Comes 1967)<br>0.43/day (calculated from Yeo 1967)<br>0.04/day (calculated from Simsiman and Chesters 1976) |
| $t_{1/2}$ | 1-2 days (estimated from Hiltbran et al. 1972) |
| Photolysis | 50% loss in 48 hr with ultraviolet radiation (Zepp et al. 1975)<br>Major process (WSSA 1983) |
| Oxidation | Stable (Kearney and Kaufman 1976)<br>Not significant at pH <9 (Zepp et al. 1975) |
| Hydrolysis | Potential alkaline hydrolysis (WSSA 1983) |
| Volatilization | Not significant (Simsiman and Chesters 1976) |
| Sediment sorption | Kp = 31.2 (calculated from Tucker et al. 1967)<br>Kp = 40.5 (calculated from Simsiman and Chesters 1976)<br>Inactivated by clay and suspended sediment (WSSA 1983) |
| Water solubility | Soluble (WSSA 1983) |
| Bioconcentration | <1 (Haven 1969) |
| Biotransformation and biodegradation | Not well known; bound diquat is apparently persistent (Summers 1980) |

* Kp is the sorption rate coefficient on sediment.

Table 6

Endothall - Herbicide Information

Common and Trade Names:

Endothall

AQUATHOL GRANULAR AQUATIC HERBICIDE (granular, 10.1%, dipotassium salt)

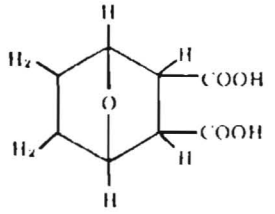
AQUATHOL K (liquid, 40.3%, dipotassium salt)

I.U.P.A.C. Name:

7-Oxabicyclo-2,2,1 heptane-2,3-dicarboxylic acid

Formula and Structure:

$C_8H_{10}O_5$




Mode of Action:

1. Contact herbicide (Ashton and Crafts 1981).
2. Causes desiccation and browning of foliage (Klingman *et al*. 1975).
3. Inhibits protein synthesis (Haller and Sutton 1973).
4. Reduces respiration (Haller and Sutton 1973).
5. Decreases lipid metabolism (Haller and Sutton 1973).

Table 7

Environmental Fate Rate Constants for Endothall

| Process | Rate |
|---|---|
| $K_T$ (overall) | 0.27/day (calculated from Hiltibran 1963)<br>0.17/day (calculated from Holmberg and Lee 1976)<br>0.095/day (calculated from Yeo 1970)<br>0.45/day (calculated from Frank and Comes 1967) |
| $t_{1/2}$ | 1 to 4 days (Rodgers et al. 1984) |
| Photolysis | Stable (Pennwalt Corp. literature) |
| Oxidation | Stable (Pennwalt Corp. literature) |
| Hydrolysis | Stable (Pennwalt Corp. literature) |
| Volatilization | Not significant |
| Sediment sorption | Kp = 2-5 (Reinert and Rodgers 1984)<br>Kp = 0.56 (calculated using Neely and Mackay 1981)<br>Kp < 1 (Reinert and Rodgers 1984) |
| Water solubility (potassium salt) | 1228 g/ℓ (Pennwalt Corp.) |
| Bioconcentration | BCF = 1.05 (Audus 1976)<br>Kow = 1.36 (calculated using Neeley and Mackay 1981) |
| Biotransformation and biodegradationon | Major process<br>$K_1 = 0.1\ d^{-1}$ (Reinert et al. in press) |

Table 8

Fluridone - Herbicide Information

Common and Trade Names:

Fluridone

SONAR AS (liquid, 45.2%)

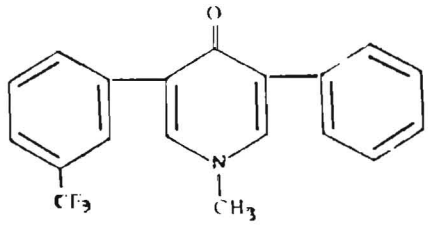
SONAR 5P (granular, 5%)

I.U.P.A.C. Name:

1-methyl-3-phenyl-5-[3-(trifluromethyl)phenyl]-4(1H)pyridinone

Formula and Structure:

$C_{19}H_{14}F_3N\ O$




Mode of Action:

1. Inhibits carotenoid synthesis (McCowen et al. 1979).
2. Promotes chlorophyll degradation due to carotenoid loss (McCowen et al. 1979).

Table 9

Environmental Fate Rate Constants for Fluridone

| Process | Rate |
|---|---|
| $K_T$ (overall) | 0.15/day (McCowen et al. 1979) |
| $t_{1/2}$ | 21-22 days persistence (Elanco Technical Report 1981)<br>4-55 days |
| Photolysis | Major process (Elanco Technical Report 1981; McCowen et al. 1979)<br>0.7 day (calculated from WSSA 1983) |
| Oxidation | -- |
| Hydrolysis | Stable (Elanco Technical Report 1981; McCowen et al. 1979) |
| Volatilization | $P < 1 \times 10^{-7}$ torr at 25°C (Elanco Technical Report 1981) |
| Sediment sorption | Kp = 3.26 (McCowen et al. 1979)<br>Strongly sorbed (WSSA 1983) |
| Water solubility | 12 mg/ℓ (Elanco Technical Report 1981) |
| Bioconcentration | BCF = 0.9-3.7 (Elanco Technical Report 1981)<br>Kow = 74.1 (Elanco Technical Report 1981) |
| Biotransformation and biodegradation | Occurs (Elanco Technical Report 1981) |

Table 10

Acute Toxicity Values for Copper*

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (μg/ℓ) | Species Mean Acute Value (μg/ℓ) |
|---|---|---|---|---|---|
| | | Freshwater Species | | | |
| Worm, *Limnodrilus hoffmelsteri* | S, U | Copper sulfate | 100 | 102 | - |
| Worm, *Nais* sp. | S, M | - | 50 | 90 | - |
| Snail (adult), *Amnicola* sp. | S, M | - | 50 | 900 | - |
| Snail, *Campeloma decisum* | FT, M | Copper sulfate | 35-55 | 1,700 | - |
| Snail, *Gyrauius circumstriatus* | S, U | Copper sulfate | 100 | 108 | - |
| Snail, *Physa heterostropha* | S, U | Copper sulfate | 100 | 69 | - |
| Snail, *Physa integra* | FT, M | Copper sulfate | 35-55 | 39 | - |
| Cladoceran, *Daphnia magna* | S, U | Copper sulfate | 226 | 200 | - |

(Continued)

* From USEPA (1980).

** S = static, FT = flow-through, R = renewal, U = unmeasured, M = measured
Results are expressed as copper, not as the compound.

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Cladoceran, *Daphnia magna* | R, U | Copper chloride | 45.3 | 9.8 | - |
| Cladoceran, *Daphnia magna* | S, U | Copper chloride | 99 | 65 | - |
| Cladoceran, *Daphnia magna* | S, U | Copper chloride | 99 | 30 | - |
| Cladoceran, *Daphnia magna* | S, U | Copper sulfate | 120 | 12.7 | - |
| Cladoceran, *Daphnia magna* | S, U | Copper sulfate | - | 100 | - |
| Cladoceran, *Daphnia magna* | S, M | Copper chloride | 52 | 26 | - |
| Cladoceran, *Daphnia magna* | S, M | Copper chloride | 105 | 30 | - |
| Cladoceran, *Daphnia magna* | S, M | Copper chloride | 106 | 38 | - |
| Cladoceran, *Daphnia magna* | S, M | Copper chloride | 201 | 69 | - |
| Cladoceran, *Daphnia magna* | S, U | Copper sulfate | 45 | 10 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Cladoceran, *Daphnia pulex* | S, U | Copper sulfate | 45 | 10 | - |
| Cladoceran, *Daphnia pulicaria* | R, M | - | 48 | 11.4 | - |
| Cladoceran, *Daphnia pulicaria* | R, M | - | 48 | 9.06 | - |
| Cladoceran, *Daphnia pulicaria* | R, M | - | 48 | 7.24 | - |
| Cladoceran, *Daphnia pulicaria* | R, M | - | 44 | 10.8 | - |
| Cladoceran, *Daphnia pulicaria* | R, M | - | 45 | 9.3 | - |
| Cladoceran, *Daphnia pulicaria* | R, M | - | 95 | 17.8 | - |
| Cladoceran, *Daphnia pulicaria* | R, M | - | 145 | 23.7 | - |
| Cladoceran, *Daphnia pulicaria* | R, M | - | 245 | 27.3 | - |
| Scud, *Gammarus pseudolimnaeus* | FT, M | Copper sulfate | 35-55 | 20 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Scud, *Gammarus* sp. | S, M | - | 50 | 910 | - |
| Crayfish, *Orconectes rusticus* | FT, M | Copper sulfate | 100-125 | 3,000 | - |
| Stonefly, *Acroneuria lycorias* | S, M | Copper sulfate | 40 | 8,300 | - |
| Damselfly, Unidentified | S, M | - | 50 | 4,600 | - |
| Midge, *Chironomus* sp. | S, M | - | 50 | 30 | - |
| Caddisfly, Unidentified | S, M | - | 50 | 6,200 | - |
| Rotifer, *Philodina acuticornis* | S, M | Copper sulfate | 40 | 160 | - |
| Rotifer, *Philodina acuticornis* | R, U | Copper sulfate | 25 | 700 | - |
| Rotifer, *Philodina acuticornis* | R, M | Copper sulfate | 81 | 1,100 | - |
| American eel, *Anguilla rostrata* | S, M | Copper nitrate | 53 | 6,400 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| American eel, *Anguilla rostrata* | S, M | - | 55 | 6,000 | - |
| Coho salmon (adult), *Oncorhynchus kisutch* | FT, M | Copper chloride | 20 | 46 | - |
| Coho salmon (yearling), *Oncorhynchus kisutch* | S, M | Copper chloride | 89-99 | 74 | - |
| Coho salmon (yearling), *Oncorhynchus kisutch* | S, M | Copper chloride | 89-99 | 70 | - |
| Coho salmon (smolt), *Oncorhynchus kisutch* | S, M | Copper chloride | 89-99 | 60 | - |
| Chinook salmon (alevin), *Oncorhynchus tshawytscha* | FT, M | - | 25 | 26 | - |
| Chinook salmon (swim-up), *Oncorhynchus tshawytscha* | FT, M | - | 25 | 19 | - |
| Chinook salmon (parr), *Oncorhynchus tshawytscha* | FT, M | - | 25 | 38 | - |
| Chinook salmon (smolt), *Oncorhynchus tshawytscha* | FT, M | - | 25 | 26 | - |
| Chinook salmon, *Oncorhynchus tshawytscha* | FT, M | - | 13 | 10 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Chinook salmon, *Oncorhynchus tshawytscha* | FT, M | - | 46 | 22 | - |
| Chinook salmon, *Oncorhynchus tshawytscha* | FT, M | - | 182 | 85 | - |
| Chinook salmon, *Oncorhynchus tshawytscha* | FT, M | - | 359 | 130 | - |
| Cutthroat trout, *Salmo clarki* | FT, M | Copper chloride | 205 | 367 | - |
| Cutthroat trout, *Salmo clarki* | FT, M | Copper chloride | 70 | 186 | - |
| Cutthroat trout, *Salmo clarki* | FT, M | Copper chloride | 18 | 36.8 | - |
| Cutthroat trout, *Salmo clarki* | FT, M | Copper chloride | 204 | 232 | - |
| Cutthroat trout, *Salmo clarki* | FT, M | Copper chloride | 83 | 162 | - |
| Cutthroat trout, *Salmo clarki* | FT, M | Copper chloride | 31 | 73.6 | - |
| Cutthroat trout, *Salmo clarki* | FT, M | Copper chloride | 160 | 91 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Cutthroat trout, *Salmo clarki* | FT, M | Copper chloride | 74 | 44.4 | - |
| Cutthroat trout, *Salmo clarki* | FT, M | Copper chloride | 26 | 15.7 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 30 | 19.9 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 32 | 22.4 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 31 | 28.9 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 31 | 30 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 30 | 30 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 101 | 176 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 101 | 40 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 99 | 33.1 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 102 | 30.7 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 101 | 46.3 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 99 | 47.9 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 100 | 48.1 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 100 | 81.1 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 98 | 85.9 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 370 | 232 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 366 | 70 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 371 | 82.2 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 361 | 298 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (μg/ℓ) | Species Mean Acute Value (μg/ℓ) |
|---|---|---|---|---|---|
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 169 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 85.3 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 83.3 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 103 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 274 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 128 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 221 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 165 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 197 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 514 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (μg/ℓ) | Species Mean Acute Value (μg/ℓ) |
|---|---|---|---|---|---|
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper chloride | 194 | 243 | - |
| Rainbow trout (alevin), *Salmo gairdneri* | FT, M | - | 25 | 28 | - |
| Rainbow trout (swim-up), *Salmo gairdneri* | FT, M | - | 25 | 17 | - |
| Rainbow trout (parr), *Salmo gairdneri* | FT, M | - | 25 | 18 | - |
| Rainbow trout (smolt), *Salmo gairdneri* | FT, M | - | 25 | 29 | - |
| Rainbow trout (adult), *Salmo gairdneri* | FT, M | Copper chloride | 42 | 57 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 350 | 102 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 125 | 200 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 125 | 200 | - |
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 125 | 190 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (μg/ℓ) | Species Mean Acute Value (μg/ℓ) |
|---|---|---|---|---|---|
| Rainbow trout, *Salmo gairdneri* | FT, M | Copper sulfate | 125 | 210 | - |
| Rainbow trout, *Salmo gairdneri* | S, M | Copper sulfate | 290 | 890 | - |
| Atlantic salmon, *Salmo salar* | FT, M | Copper sulfate | 20 | 48 | - |
| Atlantic salmon, *Salmo salar* | S, M | - | 8-10 | 125 | - |
| Atlantic salmon, *Salmo salar* | FT, M | - | 14 | 32 | - |
| Brook trout, *Salvelinus fontinalls* | FT, M | Copper sulfate | 45 | 100 | - |
| Stoneroller, *Campostoma anomalum* | FT, M | Copper sulfate | 200 | 290 | - |
| Goldfish, *Carassius auratus* | S, U | Copper sulfate | 20 | 36 | - |
| Goldfish, *Cyprinus carplo* | FT, M | Copper sulfate | 52 | 300 | - |
| Carp, *Cyprinus carplo* | S, M | Copper nitrate | 53 | 810 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Carp, *Cyprinus carplo* | S, M | - | 55 | 800 | - |
| Longfin dace, *Agosia chrysogaster* | R, M | Copper sulfate | 221 | 860 | - |
| Striped shiner, *Notropis chrysocephalus* | FT, M | Copper sulfate | 200 | 790 | - |
| Striped shiner, *Notropis chrysocephalus* | FT, M | Copper sulfate | 200 | 1,900 | - |
| Bluntnose minnow, *Pimephales notatus* | FT, M | Copper sulfate | 200 | 290 | - |
| Bluntnose minnow, *Pimephales notatus* | FT, M | Copper sulfate | 200 | 260 | - |
| Bluntnose minnow, *Pimephales notatus* | FT, M | Copper sulfate | 200 | 260 | - |
| Bluntnose minnow, *Pimephales notatus* | FT, M | Copper sulfate | 200 | 280 | - |
| Bluntnose minnow, *Pimephales notatus* | FT, M | Copper sulfate | 200 | 340 | - |
| Bluntnose minnow, *Pimephales notatus* | FT, M | Copper sulfate | 194 | 210 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Bluntnose minnow, *Pimephales notatus* | FT, M | Copper sulfate | 194 | 220 | - |
| Bluntnose minnow, *Pimephales notatus* | FT, M | Copper sulfate | 194 | 270 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | Copper sulfate | 202 | 460 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | Copper sulfate | 202 | 490 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | - | 200 | 790 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | - | 45 | 200 | - |
| Fathead minnow, *Pimephales promelas* | S, U | Copper sulfate | 360 | 1,450 (2)*** | - |
| Fathead minnow, *Pimephales promelas* | S, U | Copper sulfate | 20 | 23 (4)*** | - |
| Fathead minnow, *Pimephales promelas* | S, U | Copper sulfate | 200 | 430 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | Copper sulfate | 200 | 470 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Fathead minnow, *Pimephales promelas* | S, U | Copper sulfate | 31 | 84 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | Copper sulfate | 31 | 75 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | Copper sulfate | 200 | 440 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | Copper sulfate | 200 | 490 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | - | 48 | 114 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | - | 45 | 121 | - |
| Fathead minnow, *Pimephales promelas* | FT, M | - | 46 | 88.5 | - |
| Blacknose dace, *Rhinichthys atratulus* | FT, M | Copper sulfate | 200 | 320 | - |
| Creek chub, *Semotilus atromaculatus* | FT, M | Copper sulfate | 200 | 310 | - |
| Brown bullhead, *Ictalurus nebulosus* | FT, M | Copper sulfate | 202 | 180 (2)*** | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Brown bullhead, *Ictalurus nebulosus* | FT, M | sulfate | 200 | 540 | - |
| *Banded killifish, Fundulus diaphanus* | *S, M* | *Copper* nitrate | *53* | *860* | - |
| Banded killifish, *Fundulus diaphanus* | S, M | - | 55 | 840 | - |
| *Flagfish, Jordanella floridae* | FT, M | - | 350-375 | 1,270 | - |
| *Guppy, Poecilla reticulata* | *S, U* | *Copper* sulfate | *20* | *36* | - |
| Guppy, *Poecilla reticulata* | FT, M | - | 87.5 | 112 | - |
| *Guppy, Poecilla reticulata* | FT, M | - | 67.2 | 138 | - |
| *White perch, Morone americanus* | *S, M* | *Copper* nitrate | *53* | *6,200* | - |
| White perch, *Morone americanus* | S, M | - | 55 | 6,400 | - |
| *Striped bass, Morone saxatills* | *S, M* | *Copper* nitrate | 53 | 4,300 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Striped bass, *Morone saxatills* | S, M | - | 55 | 4,000 | - |
| Striped bass, *Morone saxatills* | S, U | Copper sulfate | 35 | 620 | - |
| Striped bass (larva), *Morone saxatills* | S, U | - | 68.4 | 50 | - |
| *Striped bass (larva), Morone saxatills* | S, U | - | 68.4 | 100 | - |
| *Striped bass (fingerling), Morone saxatills* | S, U | - | 68.4 | 150 | - |
| *Rainbow darter, Etheostoma caeruleum* | *FT, M* | *Copper* sulfate | *200* | *320* | - |
| Orangethroat darter, *Etheostoma spectabile* | FT, M | Copper sulfate | 200 | 850 | - |
| Pumpkinseed, *Lepomis gibbosus* | S, M | Copper nitrate | 53 | 2,400 | - |
| Pumpkinseed, *Lepomis gibbosus* | S, M | - | 55 | 2,700 | - |
| *Pumpkinseed, Lepomis gibbosus* | *FT, M* | *Copper* sulfate | *125* | *1,240* | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (μg/ℓ) | Species Mean Acute Value (μg/ℓ) |
|---|---|---|---|---|---|
| Pumpkinseed, *Lepomis gibbosus* | FT, M | Copper sulfate | 125 | 1,300 | - |
| Pumpkinseed, *Lepomis gibbosus* | FT, M | Copper sulfate | 125 | 1,670 | - |
| Pumpkinseed, *Lepomis gibbosus* | FT, M | Copper sulfate | 125 | 1,940 | - |
| Pumpkinseed, *Lepomis gibbosus* | FT, M | Copper sulfate | 125 | 1,240 | - |
| Pumpkinseed, *Lepomis gibbosus* | FT, M | Copper sulfate | 125 | 1,660 | - |
| Pumpkinseed, *Lepomis gibbosus* | FT, M | Copper sulfate | 125 | 1,740 | - |
| Bluegill *Lepomis macrochirus* | FT, M | Copper sulfate | 45 | 1,100 | - |
| Bluegill *Lepomis macrochirus* | FT, M | Copper sulfate | 200 | 8,300 | - |
| Bluegill *Lepomis macrochirus* | FT, M | Copper sulfate | 200 | 10,000 | - |
| Bluegill *Lepomis macrochirus* | S, U | Copper chloride | 43 | 1,250 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Bluegill, *Lepomis macrochirus* | S, U | Copper sulfate | 20 | 660 | - |
| Bluegill, *Lepomis macrochirus* | S, U | Copper sulfate | 360 | 10,200 | - |
| Bluegill, *Lepomis macrochirus* | FT, M | Copper sulfate | 35 | 2,400 | - |
| Largemouth bass, *Micropterus salmoides* | R, U | Copper nitrate | 100 | 6,970 | - |
| | | Saltwater Species | | | |
| Polychaete worm, *Neanthes arenaceodentata* | FT, M | Copper nitrate | 77 | - | - |
| Polychaete worm, *Neanthes arenaceodentata* | FT, M | Copper | - | 200 | 124 |
| Polychaete worm, *Nerels diversicolor* | S, U | Copper sulfate | - | 200 | - |
| Polychaete worm, *Nerels diversicolor* | S, U | Copper sulfate | - | 445 | - |
| Polychaete worm, *Nerels diversicolor* | S, U | Copper sulfate | - | 480 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (μg/ℓ) | Species Mean Acute Value (μg/ℓ) |
|---|---|---|---|---|---|
| Polychaete worm, *Nerels diversicolor* | S, U | Copper sulfate | - | 410 | 364 |
| Polychaete worm, *Phyllodoce maculata* | S, U | Copper sulfate | - | 120 | 120 |
| Pacific oyster, *Crassostrea gigas* | FT, M | Copper sulfate | - | 560 | 560 |
| American oyster, *Crassostrea virginica* | S, U | Copper sulfate | - | 128 | 128 |
| Black abalone, *Hallotis cracherodii* | S, U | Copper sulfate | - | 50 | 50 |
| Red abalone, *Hallotis rufescens* | S, U | Copper sulfate | - | 65 | - |
| Red abalone (larva), *Hallotis rufescens* | S, U | Copper sulfate | - | 114 | 86 |
| Soft shelled clam, *Mya arenaria* | S, U | Copper chloride | - | 39 | 39 |
| Calanoid copepod, *Acartia clausi* | S, U | Copper chloride | - | 52 | 52 |
| Calanoid copepod, *Acartia tonsa* | S, U | Copper chloride | - | 17 | - |

(Continued)

Table 10 (Continued)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (µg/ℓ) | Species Mean Acute Value (µg/ℓ) |
|---|---|---|---|---|---|
| Calanoid copepod, *Acartia tonsa* | S, U | Copper chloride | - | 55 | - |
| Calanoid copepod, *Acartia tonsa* | S, U | Copper chloride | - | 31 | 31 |
| Copepod, *Eurytemora affinis* | S, U | Copper chloride | - | 526 | 526 |
| Copepod, *Pseudodiaptomus coronatus* | S, U | Copper chloride | - | 138 | 138 |
| Copepod, *Tigriopus japonicus* | S, U | Copper chloride | - | 487 | 487 |
| Mysid shrimp, *Mysidopsis bahia* | FT, M | Copper nitrate | - | 181 | 181 |
| Mysid shrimp, *Mysidopsis bigelowi* | FT, M | Copper nitrate | - | 141 | 141 |
| American lobster (larva), *Homarus americanus* | S, U | Copper nitrate | - | 48 | - |
| American lobster (adult), *Homarus americanus* | S, U | Copper sulfate | - | 100 | 69 |
| Brown shrimp, *Crangon crangon* | S, U | Copper sulfate | - | 330 | 330 |

(Concluded)

Table 10 (Concluded)

| Species | Method** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | $LC_{50}/EC_{50}$ (μg/ℓ) | Species Mean Acute Value (μg/ℓ) |
|---|---|---|---|---|---|
| Shore crab (larva), *Carcinus maenus* | S, U | Copper sulfate | - | 600 | 600 |
| Florida pompano, *Trachinotus carolinus* | S, U | Copper sulfate | - | 360 | - |
| Florida pompano, *Trachinotus carolinus* | S, U | Copper sulfate | - | 380 | - |
| Florida pompano, *Trachinotus carolinus* | S, U | Copper sulfate | - | 510 | 412 |
| Atlantic silverside (larva), *Menidia menidia* | FT, M | Copper nitrate | - | 136 (7)*** | 136 |
| Summer flounder (embryo), *Paralichthys dentatus* | FT, M | Copper chloride | - | 28 (3)*** | 28 |
| Winter flounder (embryo), *Pseudopieuronectes americanus* | FT, M | Copper nitrate | - | 129 (9) | 129 |

(Sheet 21 of 21)

Arithmetic mean of (N) results.

NOTE: Freshwater acute toxicity vs hardness:

Cladoceran, *Daphnia magna*: slope = 1.34, Intercept = -2.64, r = 0.80, p = 0.01, N = 10
Cladoceran, *Daphnia pullcaria*: slope = 0.70, Intercept = -0.40, r = 0.94, p = 0.01, N = 8
Chinook salmon, *Oncorhynchus tshawytscha*: slope = 0.67, Intercept = 0.93, r = 0.93, p = 0.01, N = 8
Cutthroat trout, *Salmo clarki*: slope = 0.88, Intercept = 0.79, r = 0.78, p = 0.01, N = 9
Rainbow trout, *Salmo gairdneri*: slope = 0.87, Intercept = 0.33, r = 0.78, p = 0.01, N = 39
Fathead minnow, *Pimephales promelas*: slope = 1.12, Intercept = 0.38, r = 0.96, p = 0.01, N = 15
Bluegill, *Lepomis macrochirus*: slope = 1.00, Intercept = 3.60, r = 0.95, p = 0.01, N = 7
Arithmetic mean acute slope = 0.94

Table 11

Chronic Values for Copper*

| Species | Test** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | Limits (µg/ℓ) | Chronic Value (µg/ℓ) |
|---|---|---|---|---|---|
| | | Freshwater Species | | | |
| Snail, *Campeloma decisum* | LC | Copper sulfate | 45 | 8-14.8 | 10.9 |
| Snail, *Physa integra* | LC | Copper sulfate | 45 | 8-14.8 | 10.9 |
| Cladoceran, *Daphnia magna* | LC | Copper chloride | 51 | 11.4-16.3 | 13.6 |
| Cladoceran, *Daphnia magna* | LC | Copper chloride | 104 | 20-43 | 29.0 |
| Cladoceran, *Daphnia magna* | LC | Copper chloride | 211 | 7.2-12.6 | 9.5 |
| Scud, *Gammarus pseudolimnaeus* | LC | Copper sulfate | 45 | 4.6-8 | 6.1 |
| Rainbow trout, *Salmo gairdneri* | ELS | Copper sulfate | 45.4 | 11.4-31.7 | 19 |
| Brown trout, *Salmo trutta* | ELS | Copper sulfate | 45.4 | 22.0-43.2 | 30.8 |
| Brook trout, *Salvelinus fontinalls* | LC | Copper sulfate | 45 | 9.5-17.4 | 12.9 |

(Continued)

* From USEPA (1980).

** LC = life cycle or partial life cycle; ELS = early life stage
Results are expressed as copper, not as the compound.

Table 11 (Continued)

| Species | Test** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | Limits (μg/ℓ) | Chronic Value (μg/ℓ) |
|---|---|---|---|---|---|
| Brook trout, *Salvelinus fontinalis* | ELS | Copper sulfate | 45.4 | 22.3-43.5 | 31.1 |
| Brook trout, *Salvelinus fontinalis* | ELS | Copper sulfate | 37.5 | 3-5 | 3.9 |
| Brook trout, *Salvelinus fontinalis* | ELS | Copper sulfate | 187 | 5-8 | 6.3 |
| Lake trout, *Salvelinus namaycush* | ELS | Copper sulfate | 45.4 | 22.0-42.3 | 30.5 |
| Northern pike, *Esox lucius* | ELS | Copper sulfate | 45.4 | 34.9-104.4 | 60.4 |
| Bluntnose minnow, *Pimephales notatus* | LC | Copper sulfate | 194 | 4.3-18 | 8.8 |
| Fathead minnow, *Pimephales promelas* | LC | Copper sulfate | 198 | 14.5-33 | 21.9 |
| Fathead minnow, *Pimephales promelas* | LC | Copper sulfate | 30 | 10.6-18.4 | 14.0 |
| Fathead minnow, *Pimephales promelas* | LC | Copper sulfate | 200 | 24-32 | 27.7 |
| Fathead minnow, *Pimephales promelas* | ELS | - | 45 | 13.1-26.2 | 18.5 |

(Concluded)

(Sheet 2 of 3)

Table 11 (Concluded)

| Species | Test** | Chemical | Hardness (mg/ℓ as $CaCO_3$) | Limits (μg/ℓ) | Chronic Value (μg/ℓ) |
|---|---|---|---|---|---|
| White sucker, *Catostomus commersoni* | ELS | Copper sulfate | 45.4 | 12.9-33.8 | 20.9 |
| Channel catfish, *Ictalurus punctatus* | ELS | Copper sulfate | 36 | 12-18 | 14.7 |
| Channel catfish, *Ictalurus punctatus* | ELS | Copper sulfate | 186 | 13-19 | 15.7 |
| Bluegill, *Lepomis macrochirus* | LC | Copper sulfate | 45 | 21-40 | 29.0 |
| Walleye, *Stizostedion vitreum* | ELS | Copper sulfate | 35 | 13-21 | 16.5 |
| | | Saltwater Species | | | |
| Mysid shrimp, *Mysidopsis bahia* | LC | Copper nitrate | 54 | 38-77 | 54 |

Table 12

Acute Toxicity Values for the Inorganic Salts of Endothall*

(dipotassium or disodium endothall)

| Species (Reference) | Exposure Period (hours) | Conditions | $LC_{50}$ Value (mg/ℓ) |
|---|---|---|---|
| Bass | | | |
| Largemouth | 24 | Static | > 200 |
| | 48 | " | 200 |
| | 48 | " | 320 |
| | 96 | Flow Through | > 135 |
| | 96 | Static | 120-125 |
| Striped | 24 | " | 2,000 |
| | 48 | " | 1,700 |
| | 96 | " | 710 |
| Carp | | | |
| Carp-Goldfish Hybrid | 96 | Static | 145-210 |
| Catfish | | | |
| Yellow Bullhead | 96 | Static | 170-175 |
| Black Bullhead | 96 | " | 180-185 |
| Channel Catfish** | 96 | " | 150 |
| Minnows | | | |
| Bluntnose | 96 | Static | 110-120 |
| Fathead | 48 | " | 480 |
| Harlequin | 24 | " | 565 |
| | 48 | " | 460 |
| Red Shiner | 96 | " | 105 |
| Redfin Shiner | 96 | " | 95 |
| Salmonid | | | |
| Chinook | 0.5 | Static | 4,900 |
| | 24 | " | 260 |
| | 24 | " | 155 |

(Continued)

* Pennwalt Corporation (1984).
** From Johnson and Finley (1980).

Table 12 (Concluded)

| Species (Reference) | Exposure Period (hours) | Conditions | $LC_{50}$ Value (mg/ℓ) |
|---|---|---|---|
| Chinook | 48 | " | 136 |
| Coho** | 96 | " | > 100 |
| Rainbow Trout** | 96 | " | 230-450 |
| Chinook | 96 | Static | 82 |
| Sunfish | | | |
| Bluegill | 24 | Static | 428 |
| | 24 | " | 450 |
| | 24 | Static, Soft | 450 |
| | 24 | Static, Hard | 390 |
| | 24 | Static | < 800 |
| | 48 | Static | 268 |
| | 48 | " | 280 |
| | 48 | Static, Soft | 320 |
| | 48 | Static, Hard | 240 |
| | 48 | Static | > 300 |
| | 96 | " | 125-150 |
| Redear | 96 | " | 125 |

Table 13

Effects of Repeated Exposure of Fish to the Inorganic Salts of Endothall**
(dipotassium or disodium endothall)

| Species (Reference) | Exposure Period (days) | Concentration (mg/ℓ) | Results |
|---|---|---|---|
| Bass | | | |
| Largemouth | 7 | 95-115 | Minimum effect level |
| Largemouth Fry | 3 | 10-100 | 90% survival |
| Smallmouth Fry | 8 | 10- 25 | No mortality (newly hatched) |
| Unspecified | NS* | 10 | No mortality |
| | 21 | 10 | No mortality |
| Carp | | | |
| Carp-Goldfish Hybrid | 7 | 110-150 | Minimum effect level |
| Catfish | | | |
| Black Bullhead | 7 | 10-100 | 90% survival |
| Channel | 3 | 10-25 | No mortality (newly hatched) |
| Yellow Bullhead | 7 | 110-120 | Minimum effect level |
| Minnows | | | |
| Bluntnose | 21 | 40 | No mortality |
| | 7 | 70-90 | Minimum effect level |
| Fathead | NS* | 10 | No mortality |
| Red Shiner | 21 | 40 | No mortality |
| | 7 | 60 | Minimum effect level |
| Redfin Shiner | 21 | 40 | No mortality |
| | 7 | 60 | Minimum effect level |
| Salmonids | | | |
| Chinook Salmon | 14 | 10-105 | 14-day $LC_{50}$ = 62.5 mg/ℓ |
| Rainbow Trout | 21 | 10 | No mortality |
| Unspecified Salmon | 21 | 10 | No mortality |

(Continued)

* From Pennwalt Corporation (1984).
** NS = Not Specified.

Table 13 (Concluded)

| Species (Reference) | Exposure Period (days) | Concentration (mg/ℓ) | Results |
|---|---|---|---|
| Sunfish | | | |
| Bluegill | NS* | 20 | No mortality |
| | NS** | 20 | No mortality |
| | 21 | 100 | No mortality |
| | 7 | 100-105 | Minimum effect level |
| Bluegill Eggs & Fry | 8 | 10- 25 | No mortality |
| | 12 | 50-100 | No mortality |
| Bluegill Fry | 3 | 10-100 | 90% survival |
| Green Fry | 8 | 10- 25 | No mortality |
| Redear | 7 | 100 | Minimum effect level |

Table 14

Acute Toxicity Values for the Amine Salts of Endothall*

(monamine or diamine salt)

| Species (Reference) | Exposure Period (hours) | Conditions | $LC_{50}$ Value (mg/ℓ) |
|---|---|---|---|
| Bluegill Sunfish | 24 | Static | 0.8 |
| | 24 | " | 0.3** |
| | 48 | " | 0.8 |
| | 48 | " | 0.3** |
| | 96 | " | 0.06-0.2** |
| Golden Shiner | 120 | Flow Through, Soft | 1.6 |
| | 120 | Flow Through, Hard | 0.32 |
| Lake Emerald Shiner | 4 | Static | 0.75 |
| | 4 | " | 0.29** |
| | 24 | " | 0.4 |
| | 24 | " | 0.12** |
| | 48 | " | 0.35 |
| | 48 | " | 0.10** |
| | 96 | " | 0.35 |
| | 96 | " | 0.08** |
| Largemouth Bass | 96 | Static | 0.1-0.3** |
| Redear Sunfish | 96 | Static | 0.1-0.2** |
| Yellow Bullhead | 96 | Static | 0.2-0.4** |

* From Pennwalt Corporation (1984).
** Diamine salt.

Table 15

Toxicity Determinations on Aquatic Invertebrates

Exposed to the Inorganic Salts of Endothall*

(dipotassium or disodium endothall)

| Species (Reference) | Exposure Period | $LC_{50}$ Value mg/ℓ |
|---|---|---|
| *Chironomus tentans* (midge larvae) | 24 hr | 205 |
| | 72 hr | 120 |
| Clam Eggs | 48 hr | 51 |
| Clam Larvae | 12 days | 12.5 |
| *Cypretta kawatai* (ostracod) | 24 hr | 249 |
| | 72 hr | 173 |
| *Gammarus lacustris* (freshwater scud) | 96 hr | > 320 |
| | 24 hr | > 100 |
| Oyster Eggs | 48 hr | 28.2 |
| Oyster Larvae | 14 days | 48.1 |

* From Pennwalt Corporation (1984).

Table 16

Effect of Inorganic Endothall Salts on Nontarget Animals*

(dipotassium or disodium endothall)

| Organism (Reference) | Concentration mg/ℓ | Results |
|---|---|---|
| Planktonic Animals | | |
| Amphipods | 1-3 | No detrimental effects |
| Calanoida | 5 | No change in species composition or generic density |
| Cladocerans | 5 | No change in species composition or generic density |
| Cyclopoida | 5 | No change in species composition or generic density |
| Freshwater Scud | 2 | 800% increase 1st year after treatment - 300% increase in subsequent years |
| Ostracoda | 5 | Population pulse after treatment but returned to control levels |
| Benthic Animals | | |
| Beetle Larvae | 1-3 | No detrimental effects |
| Caddisfly Larvae | 5-10 | Numbers increased after treatment |
| Clams | 5-10 | Numbers increased after treatment |
| Damselfly Larvae | 1-3 | No detrimental effects |
| | 5-10 | Numbers increased after treatment |
| Dragonfly Larvae | 1-3 | No detrimental effects |
| | 5-10 | Numbers increased after treatment |
| Leeches | 5-10 | Numbers increased after treatment |
| Mayfly Nymphs | 5-10 | Numbers increased after treatment |
| | 1-3 | No detrimental effects |

(Continued)

* From Pennwalt Corporation (1984).

Table 16 (Concluded)

| Organism (Reference) | Concentration mg/ℓ | Results |
|---|---|---|
| Benthic Animals (Continued) | | |
| Midge Larvae | 1-3 | No detrimental effects |
| Midge Larvae | 5-10 | Numbers increased after treatment |
| Mosquito Larvae | 5-10 | Numbers increased after treatment |
| Oligochaetes | 5-10 | Numbers increased after treatment |
| Stoneroller Fly Larvae | 50 | 100% hatched normally |
| True Bugs | 1-3 | No detrimental effect |
| Water Bugs | 5-10 | Numbers increased after treatment |
| Littoral Animals | | |
| Beetle Adults | 1-3 | No detrimental effects |
| Crayfish | 1-3 | No detrimental effects |
| Horsefly Larvae | 5-10 | Numbers increased after treatment |
| Snails | 5-10 | Numbers increased after treatment |
| Tadpoles | 5-10 | No detrimental effects |
| Water Beetle | 5-10 | No detrimental effects |

Table 17

Acute Toxicity of Diquat to Aquatic Organisms

| Organism | Exposure Period | $LC_{50}$ Value (mg/ℓ)* | Reference |
|---|---|---|---|
| Chironomidae | 96 hr | > 100 | Wilson and Bond (1969) |
| Mayfly, *Callibaetis* sp. | 96 hr | 16.4 | Wilson and Bond (1969) |
| Caddisfly, *Limnephilus* sp. | 96 hr | 33.0 | Wilson and Bond (1969) |
| Cladoceran, *Daphnia pulex* | 8 day | 1.0 | Gilderhaus (1967) |
| Amphipod, *Hyalella azteca* | 96 hr | 0.048 | Wilson and Bond (1969) |
| Amphipod, *Gammarus fasciatus* | 96 hr (Hardwater) | > 100 | Johnson and Finley (1980) |
| Cockle, *Cardium edule* | 24 hr | > 10.0 | Portmann and Wilson (1971) |
| American oyster, *Crassostrea virginica* | 96 hr | 1.0 NTE | Butler (1965) |
| Damselfly, *Enallagma* sp. | 96 hr | > 100 | Wilson and Bond (1969) |
| Dragonfly, *Libellula* | 96 hr | > 100 | Wilson and Bond (1969) |
| White shrimp, *Penaeus setiferus* | 48 hr | 1.0 NTE | Butler (1965) |
| Sand shrimp, *Crangon crangon* | 24 hr | > 10.0 | Portmann and Wilson (1971) |
| Fathead minnow, *Pimephales promelas* | 96 hr | 10.0 NTE | Butler (1965) |
| Fathead minnow, *Pimephales promelas* | 96 hr (Softwater) | 14.0 | Surber and Pickering (1962) |

(Continued)

* Entry NTE indicates no toxic effect.

(Sheet 1 of 4)

Table 17 (Continued)

| Organism | Exposure Period | $LC_{50}$ Value (mg/ℓ)* | Reference |
|---|---|---|---|
| Fathead minnow, *Pimephales promelas* | 96 hr (Hardwater) | 14.0 | Surber and Pickering (1962) |
| Longnose killifish, *Fundulus similis* | 48 hr | 1.0 NTE | Butler (1965) |
| Goldfish, *Carassius auratus* | 96 hr | 35.0 | Gilderhaus (1967) |
| Channel catfish (fry), *Ictalurus punctatus* | 72 hr | 10.0 NTE | Jones (1965) |
| Channel catfish (adult), *Ictalurus punctatus* | 96 hr | 10.0 NTE | Lawrence et al. (1962) |
| Black bullhead(fingerling), *Ictalurus melas* | 96 hr | 170 | Johnson and Finley (1980) |
| Bluegill (fry), *L. macrochirus* | 12 day | 10.0 NTE | Hiltibran (1967) |
| Bluegill (fry), *L. macrochirus* | 72 hr | 4.0 NTE | Jones (1965) |
| Bluegill (fingerling), *L. macrochirus* | 24 hr | 525 | Hughes and Davis (1962)** |
| Bluegill (fingerling), *L. macrochirus* | 48 hr | 150 | Hughes and Davis (1962)** |
| Bluegill (fingerling), *L. macrochirus* | 96 hr | 245 | Johnson and Finley (1980) |
| Bluegill (adult), *L. macrochirus* | 96 hr | 25.0 | Gilderhaus (1967) |
| Bluegill (adult), *L. macrochirus* | 96 hr | 10.0 | Lawrence et al. (1965) |
| Bluegill (adult), *L. macrochirus* | 96 hr (Softwater) | 140 | Surber and Pickering (1962) |

(Continued)

** Cited by L. C. Folmar. 1977. Technical Paper no. 88, US Fish and Wildlife Service, US Department of the Interior, Washington, DC.

(Sheet 2 of 4)

Table 17 (Continued)

| Organism | Exposure Period | $LC_{50}$ Value (mg/ℓ)* | Reference |
|---|---|---|---|
| Bluegill (adult), *L. macrochirus* | 96 hr (Hardwater) | 140 | Surber and Pickering (1962) |
| Yellow Perch (fingerling), *Perca flavescens* | 96 hr | 60 | Johnson and Finley (1980) |
| Largemouth Bass, *Micropterus salmoides* | 72 hr | 1.0 NTE | Jones (1965) |
| Largemouth Bass, *Micropterus salmoides* | 96 hr (Softwater) | 7.8 | Surber and Pickering (1962) |
| Largemouth Bass, *Micropterus salmoides* | 48 hr | 11.0 | Muirhead-Thompson (1971)** |
| Largemouth Bass, *Micropterus salmoides* | 96 hr | 10.0 NTE | Lawrence et al. (1965) |
| Striped Bass (Larvae), *Morone saxatilis* | 24 hr | 1.0 | Hughes (1973)** |
| Striped Bass (Larvae), *Morone saxatilis* | 48 hr | 1.0 | Hughes (1973)** |
| Striped Bass (Larvae), *Morone saxatilis* | 72 hr | 1.0 | Hughes (1973)** |
| Striped Bass (Larvae), *Morone saxatilis* | 96 hr | 1.0 | Hughes (1973)** |
| Striped Bass (fingerlings), *Morone saxatilis* | 24 hr | 35.0 | Hughes (1969)** |
| Striped Bass (fingerlings), *Morone saxatilis* | 24 hr | 25.0 | Hughes (1969)** |
| Striped Bass (fingerlings), *Morone saxatilis* | 72 hr | 15.0 | Hughes (1969)** |
| Striped Bass (fingerlings), *Morone saxatilis* | 96 hr | 10.0 | Hughes (1969)** |
| Striped Bass (fingerlings), *Morone saxatilis* | 24 hr | 315 | Welborn (1969) |

(Continued)

(Sheet 3 of 4)

Table 17 (Concluded)

| Organism | Exposure Period | $LC_{50}$ Value (mg/ℓ)* | Reference |
|---|---|---|---|
| Striped Bass (fingerlings), *Morone saxatilis* | 48 hr | 155 | Welborn (1969) |
| Striped Bass (fingerlings), *Morone saxatilis* | 96 hr | 80.0 | Welborn (1969) |
| Walleye, *Stizostedion vitreum* | 96 hr | 2.1 | Gilderhaus (1967) |
| Northern Pike, *Esox lucius* | 96 hr | 16.0 | Gilderhaus (1967) |
| Chinook salmon, *Onchorynchus tshawytscha* | 48 hr | 29.0 | Muirhead-Thompson (1971)** |
| Rainbow trout, *Salmo gairdneri* | 96 hr | 5.0 NTE | Lawrence et al. (1965) |
| Rainbow trout, *Salmo gairdneri* | 96 hr | 11.2 | Gilderhaus (1967) |
| Brown trout (fingerlings), *Salmo trutta* | 96 hr | 20.4 | Johnson and Finley (1980) |

(Sheet 4 of 4)

Table 18

Considerations for Operational Recommendations of a Herbicide for *Hydrilla* Management

| Herbicide | Acceptance or Rejection Criterion |
|---|---|
| Acrolein | Generally toxic to fish, invertebrates, and other wildlife |
| Copper complexes | Concern regarding hardness and toxicity |
| Copper and Diquat | Concerns same as for copper complexes and diquat |
| Dichlobenil | 90-day water-use restriction |
| Diquat | Concern regarding suspended and settled (on plants) particulates |
| Endothall | Not registered for use in flowing waters |
| Fenac | Must be applied to vegetation after drawdown |
| Fluridone | Experimental-use herbicide; concern regarding contact time |

Table 19

Summary of Fate Information for Copper Complexes, Diquat, Endothall, and Fluridone

| Herbicide | Mode of Action | Kps (Sorption) | $t_{1/2}$ | Susceptibility to Modification | | | |
|---|---|---|---|---|---|---|---|
| | | | | pH | Hardness | Turbidity | Light |
| Copper complexes | Cellular level, electron transport inhibition | Important | Remains in system | yes | yes | yes | no |
| Diquat | Forms free radicals in cells | 30-40 Important | 1-4 days | yes | no | yes | yes |
| Endothall | Contact herbicide, disrupts membrane transport | 2-5 | 1-4 days | no | no | no | no |
| Fluridone | Inhibits carotenoid synthesis | 3-4 | 4-55 days | no | no | no | yes |

# APPENDIX A: EDB IN DIQUAT

The information contained in this appendix pertains specifically to Chapter V: Chemical Control Technology. Environmental concerns about ethylene dibromide (EDB) contents in diquat warranted enclosing correspondence from Chevron to the State of Florida involving the environmental fate and EDB content of diquat.

ENCLOSURE 1 (CHEVRON CORPORATION)

March 6, 1984

INFORMATION ABOUT

ORTHO DIQUAT AND THE EDB ISSUE

Ortho Diquat, a herbicide used in the United States and abroad for aquatic weed control, contains trace quantities of ethylene dibromide (EDB). This paper has been prepared to address questions raised about Diquat as a result of recent regulatory actions and public concern regarding EDB.

The manufacturing process of Diquat requires the use of EDB as an intermediate chemical. Although the manufacturing specification sets a maximum of 100 ppm * EDB, chemical analyses, which are run on each batch of Diquat produced, show that the product contains not more than 30 ppm (parts per million), and recent production shows levels as low as 10 ppm.

More significantly, however, is the fact that EDB levels are reduced drastically when Diquat is diluted in normal use.

The Diquat label, as registered by EPA for aquatic weed control, calls for a maximum usage rate of two gallons formulated product per surface acre of water. Assuming 30 parts per million EDB in formulated Diquat and a four foot water depth, this dilution rate would produce 0.057 parts per billion EDB in treated water (.000057 parts per million).

The recently issued federal EPA recommendations limit EDB levels to 30 ppb (parts per billion) in ready-to-eat grain products, 150 ppb in food requiring cooking, and 900 ppb in raw grain intended for human consumption. Certain states such as Florida have elected to establish the much more stringent tolerance of 0.1 part per billion, which is regarded as the minimum detectable level. Thus, the estimated 0.057 ppb EDB level in water treated with Diquat based on the above-assumption is far below federal recommended tolerances, and less than even the most stringent state-imposed standards to date.

There are additional environmental factors which lower the actual EDB level in Diquat-treated water even further. These include the high volatility of EDB, ultraviolet photodegradation, microbial degradation, evaporation, and dilution.

Diquat is Not a Major EDB Contributor

It is estimated that approximately 300,000,000 pounds of EDB are used in the United States each year. Chevron estimates that the total amount of EDB contributed by use of Diquat is approximately 50 pounds.

The major uses of EDB are as an antiknock agent in formulation of leaded gasoline, as a preplant soil treatment for nematodes, and as an insecticide.

* This specification was amended on March 9, 1984 by reducing the maximum level of EDB to 50 ppm.

(Continued)

Diquat EDB Residues Are Far Below the NCI Study Effect Level

The controversy over EDB stems from toxicology investigations indicating cancer, birth defects, and sterility occurred in laboratory animals, treated with EDB. A review of the data reveals that rats and mice receiving daily exposures of EDB either by drinking, breathing or skin absorption over 40 to 103 weeks developed various type of carcinomas. Data on reproductive effects is inconclusive at this time on a no-effect level.

The National Cancer Institute (NCI) data from a Gavage rat study demonstrated that animals receiving a dose of EDB of 40 mg/kg of body weight per day for 49 weeks (males) displayed evidence of cancer (females 61 weeks).

Utilizing the NCI data and assuming a label application rate of Diquat at two gallons/surface acre/four foot water depth, it can be calculated that the maximum dose of EDB received by a 60 and 20 kg person would be approximately 0.02 and 0.054 ug/kg/day, respectively. These EDB doses are 2,000,000 and 740,000 times lower, respectively, than the low dose of the NCI study.

It may also be noted an inhalation study conducted by Dow Chemical Company demonstrated that approximately 3 ppm is the no-effect observable level for EDB in the rat over a 13-week exposure period.

EDB is neither retained nor accumulated by the animal systems. It is rapidly metabolized and excreted. Urinary excretion is the major route of EDB elimination. Based on the very low exposure to EDB through use of Diquat, and in addition to its rapid elimination from the body, no unreasonable risks to man or the environment are expected to result from exposure to Diquat-treated water or crops, when Diquat is used in accordance with the label.

Diquat Is a Valuable Tool

Diquat is a unique and important aquatic plant management tool, especially in areas such as Florida which have acute water weed problems. Diquat is used in canals, lakes, ponds, irrigation channels and some other waterways for control of non-native weeds such as hydrilla, water hyacinths and water lettuce. These weeds, unless controlled, can reduce or destroy the value of a waterway for recreational uses such as boating, swimming and fishing; for agricultural uses such as irrigation; and for purposes such as flood control.

Diquat is a valuable tool for use in conjunction with mechanical and biological weed control methods. Its use is carefully governed and regulated by state and federal agencies. In Florida, as in most states, it is used under a permit system by certified applicators and public agency personnel who are trained and licensed to work with such chemical tools.

Since scientific research is always continuing, be sure to refer to the most current information and label available. Always use strictly in accordance with the label and with applicable state and federal regulations.

ENCLOSURE 2 (CHEVRON CORPORATION)



March 16, 1984

Dr. Stephen H. King
Health Program Office
1323 Winewood Blvd.
Tallahassee, Florida 32301

Dear Dr. King:

I was pleased to briefly meet with you at the March 9, 1984, Hearing and discuss the situation regarding ethylene dibromide (EDB) as it relates to the use of Diquat for aquatic weed control. In order to better address the questions you asked during the meeting concerning the environmental fate of EDB in surface water, its possible migration into ground water and biotransformation, I have prepared an outline of the information available on these issues and attached copies of the supporting documents and references.

ENVIRONMENTAL FATE OF EDB IN SURFACE WATER

The amount of EDB contributed to surface water by use of Diquat in aquatic weed control is quite low. As indicated in Appendices 1 and 2, 0.057 ppb EDB, the approximate average concentration would be present in water treated with Diquat at the maximum label use rate (2 gallons/acre/4'depth).

Once EDB enters surface water, decomposition and removal by means of hydrolysis, photodegradation, microbial degradation and volatilization or evaporation occurs. Volatilization appears to be a major route of removal of EDB from water. Based on the findings and calculations of Dr. Donald MacKay, University of Toronto, and Drs. B. V. Tucker and D. S. Lingenfelter, Chevron Chemical Company, approximately 50% of the EDB would evaporate in 5 ½ days from a pond one meter deep and with the wind blowing 10 miles per hour (1,2,3,4). This calculation does not take into account any reduction of EDB through photodegradation, hydrolysis or microbial degradation. Lingenfelter estimates an EDB half-life of 15 days based on his extrapolation from ethylbromide stability in water.

Castro and Belser presented evidence that microbial degradation is also instrumental in the dehalogenation of EDB (5). They found that EDB was not readily decomposed when placed in sterile mixtures of soil and water. However, when nutrients and microorganisms were added to the mixture, EDB

was converted to ethylene and $Br^-$ within two weeks. This finding is consistent with the estimate of Ehrenberg, et al., that the half-life of EDB in ground water is considerably longer due to the absence of sunlight, microbial action and volatilization (6).

Further Leinster, et al., claim that at elevated water temperatures and neutral pH, EDB hydrolyses to ethylene glycol and bromoethanol, the half-life of the reaction being 5-10 days (7).

MIGRATION OF EDB INTO GROUND WATER

We do not believe use of Diquat for aquatic weed control would result in contamination of ground water or potable water supplies. The reasons for our opinion are primarily reviewed in the attached article by Thomason and McKenry which discusses the factors affecting diffusion of chemicals, specifically EDB, through soil (8). Basically, these investigators claim that high soil moisture, organic matter, soil particle compaction and deflocculation would be instrumental in reducing or blocking the soil pore spaces. This effect, plus dilution and binding of EDB to organic matter, would decrease its rate of migration through the underlyling high moisture content soil layer of ponds.

Thus, the combination of hydrolysis, evaporation, photodegradation and microbial degradation in the water phase plus blocked soil pore spaces between the pond and ground water are factors that would prevent significant or measurable quantities of EDB from entering ground water. In addition, treatment of potable water by municipal water districts, involving charcoal or clay filtration processes may further reduce or remove any EDB.

In cases where potable water would be taken directly from the Diquat-treated water, we believe the EDB would undergo degradation as discussed above, providing that 14 days, as indicated on the product label, have elapsed since application of the herbicide.

BIOTRANSFORMATION

Results of toxicology studies indicate that EDB can be absorbed into the system through the skin contact, inhalation and ingestion. Nachtomi demonstrated that the major metabolic pathway of EDB is conjugation with glutathione forming S-(2-hydroxyethyl) glutathione and to a lesser extent S,S'-ethylelene-bis-glutathione (9). Nachtomi, et al., and Edwards, et al., also determined that when mice and rats were orally dosed with EDB, N-acetyl-S-(2-hydroxyethyl)cysteine and 8-(2-hydroxyethyl)cysteine were excreted in the urine (10, 11).

The biological half-life of EDB in laboratory animals appears to be short. Nachtomi and Alumot found that following i.v. injection of EDB in rats and chicks, the biological half-life was 2 and 12 hours, respectively (12). Edwards, et al., presented data estimating the biological half-life of $^{14}C$-labeled EDB in mice and guinea pigs to be less than 48 hours (11). Plotnick and Conner have confirmed these findings (13).

Although the biotransformation of EDB has not been studied in man, it seems reasonable, based on the available data and findings from animal investigations, that the chemical would undergo similar metabolism and rate of elimination.

In summary, we believe the small amount of EDB released in surface water, through aquatic weed control with Diquat, would not pose a risk to human health or adversely affect the environment. This opinion is based on the fact that removal and degradation of EDB occurs in water from hydrolysis, photodegradation, microbial degradation and volatilization. The half-life of this action is calculated to occur between 5½ - 15 days. In addition, migration or diffusion of EDB into ground water would be negligible due to binding to organic matter, dilution and reduction of pore space in high moisture content soil below the body of water.

Should you have any questions, please do not hesitate to contact me at (415) 231-6002 or (415) 233-3737.

Very truly yours,

J. E. Ford, Ph.D.
Supervisor, Product
Evaluation, Pesticides

JEF:kdm-16

Attachments

cc: Dr. Donald MacKay
University of Toronto

D. S. Lingenfelter )
B. V. Tucker ) Chevron Chemical Company

bcc: R. D. Cavalli
G. M. Doppelt
R. H. Foell
D. W. Jones - For your information.
J. N. Ospenson
L. R. Stelzer
Files - w/attachments

## APPENDIX 1

## EDB CONTENT OF ANNUAL AMOUNT OF DIQUAT SOLD IN U.S.

1. One gallon Diquat Water Weed Killer = 10.36 pounds
2. Each gallon of Diquat product contains 0.003% (30 ppm) EDB or 0.0003 pounds EDB/gallon.
3. 150,000 gallons of Diquat product sold in the U.S./year.
4. Amount of EDB in total annual sales of Diquat product =

   (0.0003 pounds EDB) x (150,000 gallons) =

   **47 pounds EDB**

5. Florida

   Approximately 26,000 gallons Diquat used annually for aquatic weed control.

   (0.0003 pounds EDB) x (26,000 gallons) =

   **8 pounds EDB**

## APPENDIX 2

### EDB CONTENT OF WATER TREATED WITH DIQUAT AT THE LABEL MAXIMUM USE RATE

1. Diquat Label Maximum Use Rate = 2 gallons Diquat/surface acre/4' water
2. Conversions:

   1 acre = 43,560 $ft^2$

   1 cubic foot = 28.316 l

   1 gallon = 3.785 l
3. One gallon Diquat = 10.36 pounds = 4710 gm containing 0.003% (by wt.) EDB

   = 0.14 gm EDB/gal.

   Two Gallons Diquat = 0.28 gm EDB
4. Amount of treated water =

   43,560 $ft^2$ x 4 ft = 174,240 $ft^3$

   174,240 $ft^3$ x 28.316 l = 4,933,780 l
5. Concentration of EDB in treated water =

   0.28 gm EDB/4,933,780 l

   = 0.057 ug/l or ppb

## REFERENCES

1. Tucker, B. V.: EDB-Calculated Volatilization from and Stability in Water. Letter to J. E. Ford, March 13, 1984.

2. Lingenfelter, D. S.: Estimated Half-Life of EDB in Water. Memo to J. Abell, March 7, 1984.

3. MacKay, D. and Leinonen, P. J.: Rate of Evaporation of Low-Solubility Contaminants from Water Bodies to Atmosphere. Evn. Sci. Tech. 9:1178, 1975.

4. MacKay, D. and Yeun, A. T. K.: Mass Transfer Coefficient Correlations for Volatilization of Organic Solutes from Water. Env. Sci. Tech. 17:211, 1983.

5. Castro, C. E. and Belser, N. O.: Biodehalogenation. Reductive Dehalogenation of Biocides Ethylene Dibromide, 1,2-Dibromo-3-chloropropane, and 2,3-Dibromobutane in Soil. Env. Sci. Tech. 2:779, 1968.

6. Ehrenberg, L., et al.: On the Reaction Kinetics and Mutagenic Activity of Methylating and Betahalogenoethylating Gasoline Additives. Radiat. Bot. 15:185, 1974.

7. Leinster, P., et al.: Ethylene Dibromide in Urban Air. Atmsph. Env. 12:2383, 1979.

8. Thomason, I. J. and McKenry, M. V.: Part I. Movement and Fate as Affected by Various Conditions in Several Soils. Hilgardia 42:11, 1974.

9. Nachtomi, E.: The Metabolism of Ethylene Dibromide in the Rat. The Enzymatic Reaction with Glutathion in vitro and in vivo. Biochem. Pharm. 19:2853, 1970.

10. Nachtomi, E., et al.: The Metabolism of Ethylene Dibromide in the Rat. I. Identification of the Detoxification Products in Urine. Israel J. Chem. 4:239, 1966.

11. Edwards, K., et al.: Studies with Alkylating Esters - II. Biochem. Pharm. 19:1783, 1970.

12. Nachtomi, E. and Alumot, E.: Comparison of Ethylene Dibromide and Carbon Tetrachloride Toxicity in Rats and Chicks - Blood and Liver Levels, Lipid Peroxidation. Exp. Mol. Pathol. 16:71, 1972.

13. Plotnick H. B. and Conner, W. L.: Tissue Distribution of $^{14}C$-Labeled Ethylene Dibromide in the Guinea Pig. Res. Com. Chem. Pathol. Pharm. 13:251, 1976.

ENCLOSURE 3

CHEVRON
ENVIRONMENTAL HEALTH CENTER

Chevron

3-13-84

Chevron Chemical Company

FILE COPY

| RTE | NAME | INIT | HOLE | CC |
|---|---|---|---|---|
| | J. N. OSPENSON | | | |
| | R. D. CAVALLI | | | |
| | J. T. KERKIS | | | |
| | J. K. KODAMA | | | |
| | J. A. MacGREGOR | | | |
| | J. McKINDLEY | | | |
| | J. C. SINCLAIR | | | |
| | J. E. FORD | | | |
| | | | | |
| RETURN | DRC | | INF | |

Richmond, California
March 13, 1984

**EDB - Calculated Volatilization from and Stability in Water**

110.1 Ethylene dibromide

J. E. FORD:

Dr. Donald MacKay, University of Toronto, has published on calculating rates at which low solubility compounds evaporate from water. I discussed with him on the phone his calculation for EDB. His calculations show that 50 percent of the EDB will evaporate in 5½ days from a pond 1 meter deep with a 10 mile per hour wind blowing. If pond contains organic matter or sediment for EDB adsorption, the rate of evaporation will decrease; i.e., will take longer than 5½ days for 50 percent of EDB to evaporate. Two of MacKay's publications are attached. The 1975 publication explains the equations used and the 1983 publication gives the data for EDB.

Dr. D. S. Lingenfelter, a Chevron formulation chemist, estimates the half-life of EDB in water at less than 15 days based on extrapolations from ethyl bromide stability in water. His report is attached.

B. V. Tucker

B. V. TUCKER

BVT:ca

cc: H. G. Franke
J. Abell

Attachments

1. D. MacKay and P. J. Leinonen, Rate of Evaporation of Low-Solubility Contaminants from Water Bodies to Atmosphere, Environ. Sci. & Tech., 9, 1178 (1975).

2. D. MacKay and A. T. K. Yeun, Mass Transfer Coefficient Correlations for Volatilization of Organic Solutes from Water, Environ. Sci. & Tech., 17, 211 (1983).

3. D. S. Lingenfelter, Estimated Half-Life of EDB in Water, March 7, 1984 memo to J. Abell.

| FROM (Name) | OFFICE SYMBOL | TELEPHONE NO. | RELEASER'S SIGNATURE | DTG |
|---|---|---|---|---|
| Westerhold | WESES-A | FTS 542-3860 | Clement | |
| TO (Name) Bob Boice | OFFICE SYMBOL | TELEPHONE NO. FTS 922-4710 | # PAGES 6 | PRECEDENCE |

SUBJECT

Estimated Half-Life of EDB in Water
March 7, 1984

Dr. J. Abell:

1,2-Dibromoethane

You had asked for an estimate of the half-life of ethylene dibromide (EDB) in water. After a brief (4 hour) study of the problem, I concluded that I was 80% confident that the half-life in water at room temperature was less than 15 days.

You then asked for a note describing the method used in making the estimate. The following should answer this need.

I first asked Ms. Milissa Lau to search our Chemical Abstracts computer data base for literature pertaining to the hydrolysis of EDB. This search was not successful.

While Melissa's data base extended back in time only to 1965, Chevron Research had a data base that extended back much further. However, carrying out a search at CRC would have taken more time than was available, so this approach was abandoned.

I then asked Melissa to begin a search on a related compound, ethyl bromide. Chemical principles suggest that the hydrolytic stability of ethyl bromide should be greater than ethylene dibromide. This is because the second bromine in EDB can assist the loss of the first bromine through the formation of a "bromonium ion" intermediate (somewhat similar to the "phenonium ion" intermediates studied by D. J. Cram and his associates in the 1950's).

This time Melissa's search was successful. The literature citation found was M. J. Blandamer, JACS, 103(9), 2415. The citation is provided as Attachment 1.

A study of the article showed that the rates of hydrolysis of ethyl bromide had been measured at a number of different temperatures, ranging from 58 degrees Centigrade to 90 degrees Centigrade. Using the relationship,

$$\tau = \frac{.693}{k}$$

where $\tau$ represents the half-life, and k represents the rate constant, the half-life of ethyl bromide at these temperatures was calculated.

I then asked Mr. Jim Swanson to apply his computer curve fitting techniques to the set of half-lives obtained above in order to extrapolate to a half-life at room temperature. While a number of curves were offered by the computer, we chose the one giving the best fit (Index of Determination was .999275).

Jim then prepared a graph using the curve we had chosen. This graph is shown as Attachment 2. As the graph indi-

cates, the extrapolated half-life of ethyl bromide is 15 days.

Thus, it seems reasonable to conclude that the half-life of EDB in water would be no more than the 15 days our data suggests for ethyl bromide.

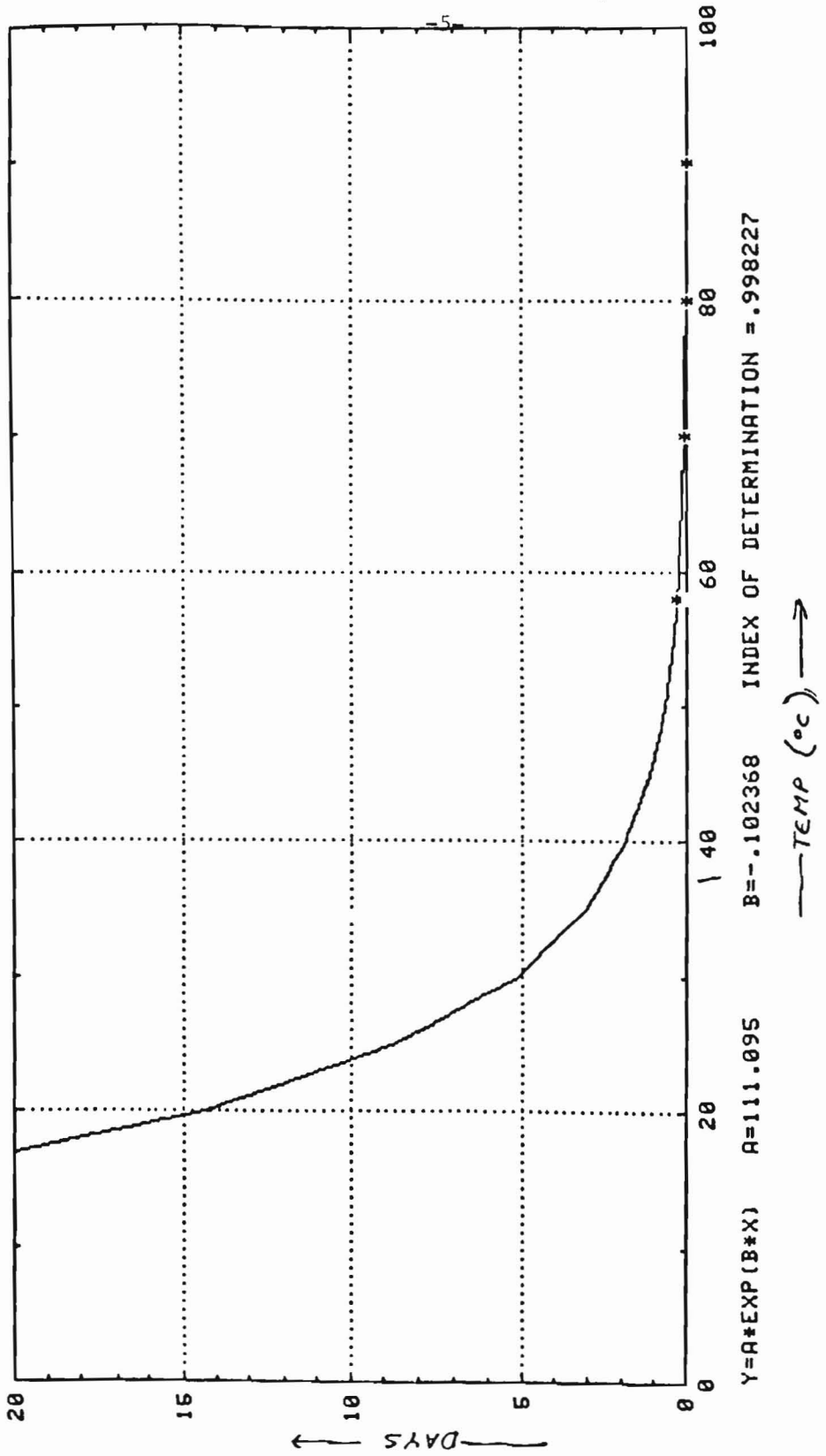
David S. Lingenfelter

Attachments
cc: Dr. B. V. Tucker

ATTACHMENT 1

ANSWER 2

AN CA94(25):208041n
TI Heat capacities of activation for displacements at primary and secondary carbon centers in water
AU Blandamer, Michael Jesse; Robertson, Ross Elmore; Golding, Peter David; MacNeil, Joseph Mark; Scott, John Marshall William
CS Chem. Dep., Univ. Leicester
LO Leicester, Engl.
SO J. Am. Chem. Soc., 103(9), 2415-16
SC 22-3 (Physical-Organic Chemistry)
DT J
CO JACSAT
IS 0002-7863
PY 1981
LA Eng
AB The rates of hydrolysis of EtBr and Me2CHOSSMe were examd. as a function of their temp. dependence with respect to 2 mechanistic models: the classical single-step mechanism and an alternate mechanism involving an intermediate. The data fit the latter model better.
KW hydrolysis ethyl bromide heat capacity; methanesulfonate hydrolysis mechanism
IT Hydrolysis
(of Et bromide and iso-Pr methanesulfonate, mechanism of, heat capacities in relation to)
IT Kinetics of hydrolysis
(of Et bromide and iso-Pr methanesulfonate, temp. dependence of)
IT Heat capacity
(of activation, for hydrolysis of Et bromide or iso-Pr methanesulfonate, mechanism in relation to)
IT 74-96-4 926-06-7
(hydrolysis of, heat capacities and mechanism for)


Y=A*EXP(B*X)
A=111.095
B=-.102368
INDEX OF DETERMINATION =.998227
DAYS
TEMP (°C)
0
5
10
15
20
0
20
40
60
80
100

DSL-3/24

Chevron

ORTHO

Chevron Chemical Company
940 Hensley Street, Richmond, CA 94804

Research and Development
Agricultural Chemicals Division

May 23, 1984

DIQUAT

Dear :

On March 28, 1984, we sent you some information about ORTHO DIQUAT Herbicide-H/A, EPA Reg. No 239-1663. The calculations regarding potential levels of EDB in water from aquatic herbicide use were based on typical EDB levels of 30 ppm in technical diquat.

Subsequent to the mailing of that package, we have been advised by our supplier, ICI Ltd., that future diquat dibromide will contain a maximum of 10 ppm EDB. This means that our calculations should be revised downward by at least one-third. For example, the maximum aquatic use rate of 2 gallons formulated product per surface acre of water, assuming 4-foot water depth and 10 ppm EDB in the product would produce an initial concentration of 0.019 ppb EDB in the treated water.

We will be filing an amended specification with EPA as soon as the required analytical documentation is completed. The first of these tests show EDB at about 7 ppm in the technical material.

Sincerely yours,

N. J. Rachman

Nancy J. Rachman, Ph.D.
Registration Specialist
and State Liasion

## APPENDIX B: COMPARISON OF CONTROL TECHNIQUES FROM THE LITERATURE

Information on the effectiveness of various control techniques was compiled by the US Army Engineer District, Jacksonville. With the exception of a small amount of mechanical control data generated by the US Army Engineer Waterways Experiment Station in the fall of 1984, the subjects of the source publications were the control of dioecious *Hydrilla* in Florida or Eurasian watermilfoil in the State of Washington or in Canada. The matrix shown in Table B1 was developed from easily accessible publications and the experience of the Jacksonville District's Aquatic Plant Control Operations Support Center (APCOSC) to compare the various potential methods of *Hydrilla* control in the Potomac River.

The ratings of general feasibility, effectiveness, control over area affected, and selectivity given in Table B1 were based on literature interpretations and experience of the APCOSC. Productivity, control cost, and duration of control were cited from the literature or taken directly from operation control programs. The long-term maintenance costs were computed by applying the single treatment cost per acre to the duration of control to maintain acceptable small boat navigation over a three-month growing period.

### References


Canellos, G. 1981. Aquatic plants and mechanical methods for their control. The MITRE Corporation, McLean, Virginia. U. S. Environmental Protection Agency contract no. 68-01-5965.

McGehee, James T. 1979. Mechanical hydrilla control in Orange Lake, Florida. Journal of Aquatic Plant Management 17:38-61.

McGehee, James T. 1984. Effects of changing regulations on aquatic plant management. Proceedings Weed Science Society of America.

Municipality of Metropolitan Seattle. 1983. Annual milfoil harvesting report. Seattle, Washington.

Perkins, Michael A. 1980. Evaluations of selected non-chemical alternatives for aquatic plant management. University of Washington, Seattle, Washington.

Province of British Columbia, Ministry of the Environment. Aquatic plant management program Okanagan Lakes. Volume IV, information bulletin. Ministry of the Environment, Water Investigations Branch.

US Army Corps of Engineers District, Jacksonville. 1985. Unpublished operational reports. Jacksonville, Florida.

US Army Corps of Engineers Waterways Experiment Station. 1984. Mechanical harvesting - Potomac River. Unpublished report. Vicksburg, Mississippi.

Table B1

Comparison of Control Techniques from the Literature

| Control Technique | General Feasibility | Productivity acre/day/work unit | Control Cost/Acre Single Treatment | Effectiveness | Duration of Control Per Treatment | Long-Term Maintenance Cost $/acre/yr | Control Over Area Affected | Selectivity |
|---|---|---|---|---|---|---|---|---|
| Biological: | | | | | | | | |
| Grass carp | Limited | | $65 | Good-excellent | 7 yr | $10 | Poor | Poor |
| Mechanical: | | | | | | | | |
| Harvester | Good | 1.3 | $484-1052* | Good | 2 wk-3 mo | $390-2880 | Excellent | Poor-fair |
| Mudcat | Fair | 0.25-4.9 | $3412* | Fair | 1-3 mo | $3411 | Excellent | Poor |
| Diver-assisted dredge | Poor/turbidity | 0.86 | $2280-2533* | Poor-excellent | 1-3 mo | $2300-2500 | Fair-excellent | Poor |
| Shoreline rototiller | Limited | 4.0 | $42-85* | Fair | 1-3 mo | $42-170 | Good-excellent | Poor |
| Rotovator | Poor-unknown | 0.5 | $776* | Fair | 1-3 mo | $800-1600 | Fair-good | Poor |
| Chemical: | | | | | | | | |
| General | Good | 4.3 | $92-614 | Good-excellent | 3-18 mo | $33-1000 | Poor-fair | Poor-good |
| Diquat | Fair-good | 4.3 | $131 | Fair-good | 2-3 mo | $262 | Fair | Fair-good |
| Copper | Limited-good | 4.3 | $92-206 | Fair-good | 2-3 mo | $184-412 | Poor-fair | Fair |
| Diquat/Copper | Good | 4.3 | $169 | Good-excellent | 2-3 mo | $338 | Fair | Fair-good |
| Aquathol K (liquid) | Good | 4.3 | $117-145 | Good-excellent | 2-3 mo | $234-290 | Poor-fair | Fair-good |
| Aquathol (granular) | Good | 4.3 | $200-475 | Good-excellent | 2-3 mo | $400-950 | Fair | Fair-good |
| Hydout | Fair | 4.3 | $151-614 | Good-excellent | 2-3 mo | $153-1228 | Poor-fair | Fair |
| Hydrothol 191 | Fair | 4.3 | $192 | Good-excellent | 2-3 mo | $384 | Poor-fair | Fair |
| Sonar AS (liquid) | Good | 4.3 | $171-307 | Good-excellent | 12-18 mo | $86-307 | Poor | Good |
| Sonar 5P (pellet) | Good | 4.3 | $177-318 | Good-excellent | 12-18 mo | $86-318 | Poor | Good |
| Bottom-covering material: | | | | | | | | |
| Hypalon | Limited | 0.5 | $16,000* | ? | 1 yr | $2166 | Excellent | Poor |
| 4-6 mil polyethylene | Limited | 0.5 | $ 4,000* | ? | 1 yr | $2166 | Excellent | Poor |
| Dartek | | | | | | | | |

* Adjusted to 1985 dollars.