



DRAFT
Supplemental Environmental
Impact Statement
Assessments of Aquatic Herbicides



July 2000
Publication Number 00-10-040



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Prepared by:
 Washington State Department of Ecology
 Water Quality Program

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Appendix D	<i>SEIS Assessments of Aquatic Herbicides</i> , Volume 2: Endothall
Appendix E	1992 SEIS Appendices: Grass Carp Supplement, Copper Compounds, Fluridone Human Health Risk Assessment, Fluridone Aquatic Risk Assessment, Glyphosate Risk Assessment, 1992 SEIS Responsiveness Summary

ACRONYMS

APMP:	Aquatic Plant Management Program
BEE:	2,4-D butoxyethyl ester (Aqua-Kleen® and Navigate®)
CWA:	Federal Water Pollution Control Act of 1972, known as the Clean Water Act
DNR:	Washington State Department of Natural Resources
EEC:	Expected Environmental Effects Concentration
EIS:	Environmental Impact Statement
EPA:	United States Environmental Protection Agency
ESA:	The Endangered Species Act
EUP:	Experimental Use Permit
FEIS:	Final Environmental Impact Statement
FIFRA:	Federal Insecticide, Fungicide, and Rodenticide Act, as amended
GMA:	Growth Management Act
HPA:	Habitat Conservation Plan (ESA Sections 10, 16 and 1539)
HPA:	Hydraulic Project Approval
IPM:	Integrated Pest Management (IPM Law is Chapter 17.15 RCW)
IAVMP:	Citizen's Manual for Developing Integrated Aquatic Vegetation Management Plans
IVMP:	Integrated Aquatic Vegetation Management Plans
LC50:	Lethal Concentration is 50%. The quantity of substance needed to kill 50% of test animals exposed to it within a specified time. This test applies to gasses, vapors, fumes and dusts.
MC:	Mosquito Control Policy
MOS:	Margin of Safety
NMFS:	National Marine Fisheries Services
NOAA:	National Oceanic and Atmospheric Association
NOEC:	No Observable Effect Concentration
NOEL:	No Observable Effect Level
NWIFC:	Northwest Indian Fisheries Commission
RCW:	Revised Code of Washington
RQ:	Risk Quotients
SEIS:	Supplemental Environmental Impact Statement
SEPA:	State Environmental Policy Act
STM:	Short-term modification of WQS, a permit per 173-201A-110 WAC
U.S.C.:	United States Code
WAC:	Washington Administrative Code
WDFW:	Washington State Department of Fish and Wildlife
WQS:	Water Quality Standards, Chapter 173-201A WAC
WSDA:	Washington State Department of Agriculture
WSU:	Washington State University

FACT SHEET

Project Title: State of Washington Aquatic Plant Management Program

Proposed Action: The Proposed Action is the application of herbicides for aquatic plant management. The action is defined as a nonproject proposal under State Environmental Policy Act (SEPA) rules such that the Environmental Impact Statement (EIS) will be integrated with on-going agency planning and permitting procedures for aquatic herbicides. The recommended alternative is an integrated aquatic plant management approach using the most appropriate mix of vegetation control methods that may include biological, manual/mechanical, and chemical methods. Also included in the preferred action is the policy supporting an integrated management approach in Ecology's permitting program. Other alternatives analyzed in this SEIS include chemical use only, mechanical use only, biological use only, and no action, which is the continuation of current policy.

Lead Agency: Washington State Department of Ecology

Responsible Official: Megan White, Water Quality Program Manager

Contact Person: Kathleen Emmett, Water Quality Program

Licenses, Permits: This list reflects permits required for various plant management alternatives discussed in this document, including use of aquatic herbicides, rotovation, dredging, manual and biological control methods. Not all permits listed below are required for all activities discussed in this document. Requirements may change; please check with resource agencies to determine permit requirements for a particular project. An overview of state programs for aquatic pesticide regulation is provided in Section I.

Ecology: Temporary Modification of Water Quality Standards

Fish and Wildlife: Hydraulic Project Approval
Fish Planting Permit

Local: Substantial Development Permit (Shorelines Management Act) in certain locales

Federal: Section 404 Permit from the Army Corps of Engineers

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SUMMARY

Aquatic plants are a valuable component of aquatic ecosystems that in normal situations require protection. Like algae, aquatic plants are a vital part of a watershed system because they provide cover, habitat and food for many species of aquatic biota, fish and wildlife. Aquatic plants also limit certain lake uses. Too many rooted and floating plants can degrade water quality, impair certain fisheries, block intakes that supply water for domestic or agricultural purposes, and interfere with navigation, recreation and aesthetics. In addition, noxious aquatic plant species such as Eurasian watermilfoil can form dense populations that may pose safety problems for swimmers and boaters and can degrade wildlife habitat by out-competing native species or changing water chemistry. Consequently, Ecology's Water Quality Program receives requests for permits from various entities to use herbicides and other control methods to manage excessive native and noxious aquatic plant species and algae in various waterbodies. In response to these requests and in accordance with the provisions of the State Environmental Policy Act (SEPA), Ecology determined that aquatic plant management by these methods may have significant adverse environmental impacts, therefore state law requires an Environmental Impact Statement (EIS).

The State of Washington Water Pollution Control Act (RCW 90.48) and the State Surface Water Quality Standards (Chapter 173-201A WAC) require the Department of Ecology (Ecology) to establish criteria and programs necessary to protect waters of the state. These standards articulate an intent to protect public health and maintain beneficial uses of surface waters, including activities such as swimming, boating, and aesthetic enjoyment; public water supply; stock watering; fish and shellfish rearing, spawning, and harvesting; wildlife habitat, and commerce and navigation. Water Quality Standards (WQS) specifically allow Ecology to modify water quality criteria on a short-term basis to accommodate essential activities, respond to emergencies, or otherwise protect the public interest.

In 1980, Ecology completed an *Environmental Impact Statement* (EIS) as guidance for a statewide Aquatic Plant Management Program issuing short-term modifications for aquatic plant control. Ecology uses this document as guidance to decide whether to approve, deny, or add conditions to permits related to aquatic plant management. The EIS evaluated the impacts of aquatic herbicides used for control of nuisance aquatic vegetation, including endothall, diquat, dichlobenil (2,6-dichlorobenzonitrile), 2,4-D [(2,4-dichlorophenoxy) acetic acid], copper sulfate, komeen and simazine. However, non-chemical control methods were not evaluated.

Since 1980, diquat, dichlobenil, 2,4-D, and simazine were discontinued for use in the program and fluridone and glyphosate were introduced. A number of mechanical and physical methods (i.e. mechanical harvesting, rotovation, bottom barriers, and cutters) have been developed and used extensively for aquatic vegetation control, and various methods of biological control have undergone research and development during the past two decades. In addition, growing concern with the impacts aquatic herbicides may have on human and environmental health resulted in new regulations to control their use. Changes also occurred in our understanding of aquatic ecosystems, including the role of wetlands and the need to consider and control nutrient and sediment loading within the total watershed of any particular waterbody.

To address these changes in aquatic plant management, Ecology updated and supplemented the EIS with the *Final Supplemental Environmental Impact Statement for the Aquatic Plant Management Program* (SEIS), dated January 1992, to examine an array of control alternatives and propose mitigation measures for significant adverse impacts. The SEIS proposed an integrated management approach as the preferred alternative but also considered the use of chemical controls only, physical controls only, biological controls only, and taking no action relative to controlling nuisance aquatic plants.

The 1992 SEIS evaluated the use of copper, endothall, fluridone and glyphosate to control various types of aquatic plants and encouraged the use of the most efficient and effective combination of control methods that minimized impacts to human or environmental health. The evaluation confirmed that having a variety of control methods available provides the flexibility necessary to control nuisance populations of invasive and non-native species in situations where it is desirable to maintain other, often-conflicting beneficial water uses. However, an integrated aquatic plant management approach takes more than a one-season planning effort by lake managers.

Currently Ecology is encouraging lake and watershed management planning to include nutrient and sediment enrichment and long-term aquatic plant control. In addition, the lake initiative segment of Ecology's Water Quality Program's Strategic Plan for sustainability projects calls for an emphasis on lake and watershed management planning to address nutrient and sediment enrichment and a de-emphasis on the use of chemicals for pest control. Ecology recommends the development of lake or aquatic plant management plans by communities or groups proposing aquatic plant control activities with participation in planning by other interested parties. Eventually, Ecology plans to issue permits for pesticide use for native plant control and algae only to areas that have an approved integrated aquatic plant management plan. An added benefit will be that these permits could be valid for up to five years, as opposed to the current one-year term of the water quality modifications.

New infestations of non-native, noxious and invasive plants often need immediate attention and should not be subject to new planning requirements the first season of treatment. In these cases, early treatment with bottom barriers, diver pulling or herbicides provides the most effective, cost-efficient and environmentally benign action to take. The Washington Legislature recognized the need for swift response to early infestations and recent legislation enables quick action to be taken in these instances (Engrossed Second Substitute Senate Bill 5633, 1996 and Senate Substitute Bill 5424, 1999). Some preventative action plans for new infestations have been developed to allow the quickest and most effective actions to be taken. Contact Ecology's Aquatic Weed Grant Program for limited funding and guidelines for early infestations in public lakes and infections of hydrilla (<http://www.wa.gov/ecology/wq/plants/grants/chapter5.html>).

Even under an integrated management program, unavoidable, significant adverse impacts may occur that will restrict other beneficial water uses. The development of a lake or aquatic plant management plan allows for the establishment of use priorities by the parties involved with maintaining and protecting the uses of a particular waterbody. Management plans help to ensure that proven control methods will be implemented for the long-term management of the waterbody and that problems such as nutrient enrichment and sediment loading, often the cause of accelerated plant and algae growth, are addressed. Planning further assures that aquatic plant managers will not rely on aquatic plant control methods that may only address the symptoms of such problems.

New chemical control methods for aquatic plants continue to evolve. In order to assess the use of new or improved products in Washington State, the 1999 Legislature directed Ecology to expand certain chemical application sections of the 1992 SEIS. The Legislature also directed Ecology to make it more responsive to the application of new, commercially available herbicides, and to evaluate their use with the most recent research available (Engrossed Substitute Senate Bill 5424, effective May 10, 1999).

To accomplish this task, Ecology initiated a SEPA environmental review process to supplement the 1992 SEIS. Ecology is the primary lead for supplemental updates to the SEIS; however, a steering committee comprised agencies with jurisdiction and/or interest in aquatic plant control provides close advisory and review assistance to the update process. Those Washington State agencies include the Departments of Agriculture, Health, Fish and Wildlife, Natural Resources, and the State Noxious Weed Control Board. The Department of Agriculture (WSDA) is charged with regulating pesticide applicators, registering pesticides

for use in the state, and, along with the State Noxious Weed Control Board, with controlling noxious plants within the state. The Department of Health (DOH) is charged with protection of human health. The Department of Fish and Wildlife (WSFW) receives requests for Hydraulic Project Approvals (HPA's) to implement various physical and mechanical methods and is charged with protecting fish and wildlife. The Departments of Natural Resources (DNR) and Ecology have concerns with the potential impact of various plant control methods on the natural resources they are charged with managing, and WSFW and DNR are under mandate to develop programs for controlling particular noxious emergent species on state-owned or managed lands.

A technical advisory committee also serves in a review capacity for the risk assessments and updates to the SEIS. The technical advisory committee enlists representatives of Lake Management Districts, local governments, scientists, tribes, pesticide registrants, and environmental groups. An external list of reviewers has also been developed for a targeted review of the new draft documents. The list of reviewers includes representatives from the Washington Legislature, the United States Environmental Protection Agency (EPA), Washington State University, National Marine Fisheries Services, National Oceanic and Atmospheric Administration, U.S. Fish & Wildlife, Northwest Coalition for Alternatives to Pesticides, Washington Toxics Coalition and the Northwest Indian Fisheries Commission.

In the fall of 1999, Ecology contracted the development of risk assessments regarding use of specific herbicide applications to provide technical support for the updates to the 1992 SEIS. The risk assessments examine the characterization and environmental fate as well as the environmental and human health effects of six herbicides. The first set of risk assessments, completed May 2000, evaluated 2,4-D formulations registered for aquatic use by the state and endothall formulations of Hydrothol 191 and Aquathol. The second set of assessments, scheduled for completion February 2001, will evaluate diquat, triclopyr, and copper compounds.

The assessments provide information for the environmental checklist required by SEPA. Application conditions that minimize or mitigate adverse human health and environmental impacts are explored, and in some cases mitigating conditions (i.e. swimming restrictions on endothall) have changed from those in the 1992 SEIS to reflect new information concerning the impacts of the product. The herbicides assessed were selected by the Agency Steering Committee for Update of the 1992 Aquatic Plant SEIS on the basis of registration status, desirability for use and direction from Senate Substitute Bill 5424 (1999).

Special consideration is given to salmonids and other listed species under the Endangered Species Act (ESA). There are several species of native salmonids in Washington, which include salmon, trout and whitefishes. Each species is comprised of many stocks and populations that vary from one another in their genetic makeup, life history and other characteristics. The National Marine Fisheries Service (NMFS) uses the concept of “evolutionary significant units” or “ESUs” to refer to any distinct group of salmon populations and to further clarify the meaning of subspecies under the Endangered Species Act (ESA). Similarly, the U.S. Fish and Wildlife Service (USFWS) refers to “distinct population segments” for species under their jurisdiction. Native salmonids in Washington that have been listed, or are proposed for listing, include:

- Chinook, commonly referred to as king salmon,
- Coho,
- Chum,
- Sockeye,

- Steelhead, an anadromous form of rainbow trout, belong to the same scientific genus as other Pacific salmon and coastal cutthroat trout,
- Coastal Cutthroat Trout, and
- Bull Trout

The risk assessments examine the potential acute and chronic effects of single and seasonally reoccurring applications on aquatic plants and animals (invertebrates and vertebrates, and associated wildlife), including consideration of life cycles and food chain impacts. Where available, information on potential impacts and toxicity of one-time and repeated applications of each herbicide on numbers, diversity, and habitat of species of plants, fish, birds and other wildlife is included. Impacts (both risks and benefits) for spawning and rearing habitat used by various species, including but not limited to fresh water trout and sea run cutthroat trout are also considered. Discussions include direct and indirect impacts of herbicide treatments on the marine environment, salmonid smoltification and their survival life histories.

Ecology's Aquatic Plant Management Program requires that permits be processed or denied depending on the potential impact to ESA listed species and other potentially affected biota, the seriousness of the aquatic plant problem and the degree to which integrated aquatic plant management plans have been considered. Also essential is conformance to the Governor of Washington's goal of no net loss of wetland functional value or acreage. Therefore each alternative must be evaluated to determine the degree to which wetlands would be impacted consistent with policies and standards being developed by Ecology and other agencies. Within this context, a priority is given to the control of non-native noxious aquatic plant species. Use of an integrated management approach will further this goal through the selection of the control method or combination of methods that will yield maximum aquatic plant control while minimizing undesirable impacts to human and environmental health.

Section I. Introduction to Lake and Aquatic Plant Management

A. Background

The state of Washington has an abundance of surface water resources, including approximately 7,800 lakes, ponds and reservoirs, 40,492 miles of rivers and streams, and untold acres of wetlands. Within these diverse waters, there is a great range of conditions such as hardness, pH, dissolved oxygen, turbidity, nutrients, size, flow, biota and use. Citizens rely on these waterbodies for a number of uses, such as recreation in the form of swimming, fishing, boating and aesthetic enjoyment; commerce and navigation; water supply for domestic, industrial and agriculture activities; and habitat for fish and wildlife.

Our understanding of how aquatic systems function has also continued to increase during the past two decades. Aquatic systems change slowly through a natural aging process called eutrophication. This process is typified by increased productivity, structural simplification of biotic components, and a reduction in the metabolic ability of organisms to adapt growth responses to imposed changes (i.e., reduced stability) (Wetzel 1975). At advanced stages of eutrophication aquatic systems are out of equilibrium with respect to the freshwater chemical and biotic characteristics desired by humans for specific purposes.

Human activities are often responsible for the introduction of exotic species that degrade aquatic environments and require extensive control measures. Human activities also affect drainage basins, water budgets, and nutrient budgets, resulting in accelerated productivity and eutrophication. As Vallentyne described (1974), a common result of misuse of the drainage basin and the excessive loading of nutrients and sediments in fresh waters is the acceleration of eutrophication, literally turning lakes into "algal bowls" (Wetzel 1975). Accelerated eutrophication often results in increased primary productivity, including increased plant growth in shallow areas of the lake. Thus, the effective treatment of excessive aquatic plant populations, including algae, must consider controlling the introduction of nutrients and sediments from sources throughout the entire watershed. Our increased knowledge of the value and function of wetlands has resulted in a reassessment of management strategies for native versus invasive species. Wetlands and native species are both usually needed to enhance or maintain aquatic systems.

B. Goals of the 1980 Environmental Impact Statement and Supplements

The 1980 EIS addressed control of aquatic plants through the use of herbicides and examined the alternative of no action. This approach treated the symptoms but not the underlying problems of lake enrichment and aquatic plant and algae growth. The 1987 amendments to the Federal Clean Water Act required the development and implementation of programs designed to reduce or eliminate the introduction of toxic substances to our nation's waters. In addition, new scientific evidence concerning the potential impacts that certain toxic substances may have on human and aquatic life have increased public awareness regarding the intentional introduction of toxic substances to surface waters, even in situations where their introduction may enhance the uses of a waterbody. Thus, a more thorough review and analysis of the benefits of aquatic herbicides relative to the potential risks to human and environmental health was deemed warranted.

Subsequently, the 1992 SEIS proposed an aquatic plant management approach that integrated herbicide use with manual, mechanical and biological methods and considered the context of whole lake and/or watershed systems.

Ecology's aquatic plant management program encourages an understanding of natural aquatic processes, including the role of aquatic plants in a natural system, plant identification and the underlying causes of excessive plant growth. Through this process, people can make informed selections of methods for reducing nutrient and sediment loading and meeting long-term management goals. This is consistent with Ecology's sustainability goals, which recommend the development of integrated aquatic plant management plans by communities, professional herbicide applicators, groups and others who request permits for aquatic plant management. Ideally, an aquatic plant management plan should be prepared before certain permits are issued for use of herbicides, and in regard to public waters, a wide range of participation is essential for the benefit of all users, not simply the adjacent property owners. However, in the case of new infestations of noxious (non-native) and invasive plants, early control may preclude the development of a plan for the first season of treatment.

Addressing the potential loss of habitat or habitat disruption from aquatic plant control strategies must also be considered as a goal in the development and implementation of any aquatic plant management program. This is especially true now that species of salmon, trout, char or steelhead have been listed in nearly every county in Washington as a candidate, a threatened or endangered species under the Endangered Species Act (ESA). Currently, Washington has 28 state candidate fish species and 3 state sensitive species including many species of marine fish. (For current listings see <http://www.governor.wa.gov/esa/regions.htm>.)

Wetlands have also often been overlooked as a key component of aquatic systems. The functional value of wetlands must be incorporated into any comprehensive lake or vegetation management plan. The Governor of Washington has adopted through executive orders a goal of no net loss of wetland functional value or acreage in the state. All management strategies for aquatic vegetation must consider this goal.

C. Aquatic Plant Control Regulation

1. Introduction

The state of Washington regulates aquatic plant control through several agencies concerned with various aspects of aquatic plant growth and control. Aquatic plants appear in many shapes and sizes. Some have leaves that float on the water surface, while others grow completely underwater. They grow wherever water is persistent, in rivers, streams, lakes, wetlands, coastlands or marine waters. In moderation, aquatic plants are aesthetically pleasing and desirable environmentally. The presence of native species is natural and normal in lakes and other water bodies because they provide important links in aquatic life systems. In large quantities, however, plants can interfere with water uses and may be seen as a problem. An over-abundance of native plants usually indicates excessive nutrients (nitrogen or phosphorus) in the water column. Conversely, non-native aquatic plants and excessive plant nutrients are often a threat to the health of the aquatic environment. The introduction of non-native aquatic plants and excessive plant nutrients has created many aquatic problems for Washington waters. The removal of non-native aquatic plants is often desirable and even necessary to enhance water quality and protect beneficial uses.

The management of aquatic plants under their respective jurisdictional authorities can be generally categorized by the control method used and by the type of plant controlled. In any case of uncertainty, the **Permit Assistance Center should be contacted at (360) 407-7037** before an aquatic plant removal or control project is initiated.

2. Regulatory Requirements for Manual, Mechanical and Biological Methods

Manual Methods The Washington State Department of Fish and Wildlife (WDFW) requires either an individual or general permit called an **Hydraulic Project Approval (HPA)** for all activities taking place in the water including hand pulling, raking and cutting of aquatic plants. However, projects conducted for the control of spartina and purple loosestrife may not require an HPA. Information regarding HPA permits can be obtained from the local office of WDFW. To request a copy of the Aquatic Plants and Fish pamphlet, please contact:

WDFW
Habitat and Lands Program
600 Capitol Way N
Olympia WA 98501-1091
(360) 902-2534 <http://www.wa.gov/wdfw/hab/aquaplnt/aquaplnt.htm>

Mechanical Cutting Mechanical cutting requires an HPA, obtained free of charge from WDFW. For projects costing over \$2,500, check with the local city or county to see if a shoreline permit is required.

Bottom Screening Bottom screening in Washington requires an HPA, obtained free from WDFW. Check with the local city or county to determine whether a shoreline permit is required.

Weed Rolling Installation of weed rolling devices requires an HPA, obtained free from WDFW. Check with your city or county to determine whether a shoreline permit is required.

Grass Carp and other Biological Controls A grass carp fish planting permit must be obtained from the WDFW, check with your regional office. Also, if inlets or outlets need to be screened, an HPA application must be completed for the screening project.

Diver Dredging Diver dredging requires an HPA from WDFW and a permit from Ecology. Check with your city or county for any local requirements before proceeding with a diver-dredging project. Also diver dredging may require a Section 404 permit from the U.S. Army Corps of Engineers.

Water Level Drawdown Permits are required for many types of projects in lakes and streams. Check with city, county and state agencies before proceeding with a water level drawdown.

Mechanical Harvesting Harvesting in Washington requires an HPA from WDFW. Some Shoreline Master Programs may also require permits for harvesting. Check with your city or county government.

Rotovation Rotovation requires several permits, including 1) An HPA from WDFW, 2) A permit from an Ecology regional office, 3) A shoreline permit from the city or county may also be needed, and 4) A Section 404 permit obtained from the Army Corps of Engineers may be required.

3. Regulatory Requirements for Aquatic Herbicide Applications

Ecology utilizes a permit system based primarily on SEPA guidance documents for implementing the requirements of the Water Quality Standards (WQS). A short-term modification (permit) may be issued by Ecology to an individual or entity proposing the aquatic application of pesticides, including but not limited to those used for control of federally or state listed noxious and invasive species, and excess populations of native aquatic plants, mosquitoes, burrowing shrimp, and fish.

Ecology is the primary lead for regulating pesticides used in aquatic environments under Washington State's Water Pollution Control Law, Chapter 90.48 RCW. However, the State Departments of Agriculture, Health, Fish and Wildlife, Natural Resources, and the State Noxious Weed Control Board are agencies with jurisdiction and/or interest in aquatic plant control.

Laws and Codes Several sections of the State Water Pollution Control Law and the WQS found in Washington's Administrative Code apply directly to the use of aquatic pesticides, including the following:

- **Definition of Waters of the State** (Chapter 90.48.020 RCW) "...lakes, rivers, ponds, inland waters, saltwater, wetlands and all other surface waters and water courses." Under this definition the requirements contained in the WQS apply to the use of aquatic pesticides in all waters of the state.
- **Toxic Substances** (WAC 173-201A-040) "Toxic substances shall not be introduced above natural background levels in waters of the state which have the potential either singularly or cumulatively to adversely affect characteristic water uses, cause acute or chronic toxicity to the most sensitive aquatic biota dependent on those waters, or adversely affect public health, as determined by the department." This requirement is consistent with CWA requirements that state WQS contain a narrative "no toxics in toxic amounts" criteria. This statement would generally prohibit the use of aquatic pesticides, so the following condition has been adopted.
- **Short-term Modifications** (WAC 173-201A-110) "... {standards} may be modified for a specific water body on a short-term basis when necessary to accommodate essential activities, respond to emergencies, or to otherwise protect the public interest even though such activities may result in a temporary reduction of water quality criteria below those criteria and classifications established by this regulation." Ecology may authorize a longer duration where the activity is part of an ongoing or long-term operation and maintenance plan, integrated pest or noxious weed management plan, waterbody or watershed management plan, or restoration plan. Such a plan must be developed through a public involvement process...and be in compliance with SEPA...in which case the standards may be modified for the duration of the plan, or for five years, whichever is less. However, a short-term modification, or permit, will only be issued if the following condition is satisfied.
- **Protection Criteria** (WAC 173-201A-110) "Such activities must be conditioned, timed, and restricted (i.e. hours or days rather than weeks or months) in a manner that will minimize water quality degradation to existing and characteristic uses. In no case will any degradation of water quality be allowed if this degradation interferes with or becomes injurious to characteristic water uses or causes long-term harm to the environment." This last statement is consistent with CWA requirements that the WQS contain an anti-degradation policy statement. Under Washington's WQS, WAC 173-201A-030 (5) beneficial uses that must be protected include fish and shellfish rearing and harvesting; salmonid and other fish migration, rearing, spawning and harvesting;

swimming; boating; navigation; irrigation; wildlife habitat; and domestic, industrial, and agricultural water supply, commerce and navigation.

EIS Guidance In 1980, Ecology completed a Final Environmental Impact Statement (FEIS) for statewide Aquatic Plant Management based primarily on aquatic herbicide use. The 1992 *Aquatic Plant Management Program's Final Supplemental Environmental Impact Statement* (Hardy, et al. 1992) updated the EIS and Ecology regional offices currently issue site-specific permits for the use of the aquatic herbicides based on this guidance. This current update effort provides guidance for aquatic formulations of Aquathol, Hydrothol 191, 2,4-D, diquat, glyphosate, fluridone and copper compounds. Copper is generally used for the control of algae. Endothall and fluridone control freshwater submersed plants such as pondweeds, elodea, hydrilla and milfoil. Glyphosate controls freshwater emergent plants such as cattails and purple loosestrife. Ecology's regional offices also issue site specific permits for the use of glyphosate on non-native, marine cordgrass (spartina) and other emergents.

Through our permitting program, Ecology encourages the use of an integrated management plan that includes the selection, integration, and implementation of proven control methods based on predicted economic, ecological, and sociological consequences. This concept is based on the premise that, in many cases, no single control method will by itself be totally successful. Thus, a variety of biological, physical, and chemical control and habitat modification techniques are integrated into a cohesive plan developed to provide long-term vegetation control. Integrated management also includes various land-use practices necessary to reduce or eliminate the introduction of nutrients and sediments that may be the cause of accelerated aquatic plant and/or algae growth. The ultimate objective is to control detrimental vegetation in an economically efficient and environmentally sound manner.

Ecology issues an annual, statewide permit to the Washington State Department of Agriculture (WSDA) based on guidance provided by the 1993 *Noxious Emergent Plant Management Environmental Impact Statement* (Ebasco 1993). This EIS is still used as Ecology's primary guidance document for permitting the use of herbicides to control noxious emergent weeds in Washington State. Upon request, WSDA provides copies of the permit to licensed applicators (with aquatic endorsements) for the use of glyphosate and 2,4-D to control the following emergent noxious aquatic plants: purple loosestrife, garden loosestrife, wand loosestrife, Japanese knotweed, indigo-bush, meadow knapweed, saltcedar and reed canarygrass. Each licensed applicator must follow the requirements of the permit. For further details, contact the WSDA Weed Specialist in Yakima at glaubrich@agr.wa.gov or (509) 225-2604.

ESA Considerations Several salmon populations and other aquatic biota are listed for special protection under the Endangered Species Act (ESA). Listings may affect aquatic control projects all over Washington State. Information regarding potential listings of endangered species in particular water bodies can be obtained from the local office of the Washington Department of Fish and Wildlife or on their website at: <http://www.governor.wa.gov/esa/regions.htm> and <http://www.wa.gov.wdfw/wlm/diversity/soc/etsc9907> .

Obtaining a permit from Ecology for the application of herbicides does not exempt an applicator from "Take" liability under ESA. "Take" means to "harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in such conduct" with respect to a species listed under ESA (16 U.S.C. Section 1532(19)). Current permit applications require applicators to state whether the waterbody proposed for treatment is part of a designated critical habitat of an ESA listed species or if the waterbody is in an Evolutionary Significant Unit listed under ESA. Proposed treatments that may have an adverse impact on a listed species may be denied or restricted for their protection.

At present, Ecology is working with NMFS, USFWS, WDFW, WSDA and the Environmental Protection Agency (EPA) to have the aquatic permitting program protected from “Take” liability under the exemption provision of the ESA 4(d) rule. A pesticide/ESA technical group and a separate policy group, both comprised of representatives from these agencies, have been meeting to review the potential risks that permitted aquatic pesticides may pose to salmonids and evaluate whether the aquatic pesticide permitting program provides adequate protection for listed species.

The NMFS science center and USFWS staffs are very satisfied with seawater challenge tests that indicate an appropriate margin of safety and will likely support the permitted use of aquatic pesticides that pass this test. At present, acceptable seawater challenge information exists for endothall, Hydrothol 191, 2,4-D and diquat. Seawater challenge tests have raised significant concerns regarding the use of copper compounds in salmonid waters. Product manufacturers will need to do these tests if they expect coverage.

Aquatic Herbicide Special Use Legislation During the 1999 legislative session, the Washington Legislature passed Engrossed Substitute Senate Bill 5424 (1999) adding a new section to 90.48 RCW. This legislation allows the use of 2,4-D without a permit within certain limitations. Only government agencies may use 2,4-D to treat Eurasian milfoil infestations without obtaining a permit if the infestation is either recently documented or remaining after the application of other control measures and is limited to twenty percent or less of the littoral zone of the lake. The agency applying the herbicide must notify the general public and lakeside residents of application events and any restrictions pursuant to the label.

A Recent WQP Policy limits the use of copper in salmon-bearing waters Given the known toxicity of copper compounds to aquatic life, primarily amphibians and fish, and given the recent ESA listings of several native salmonid species in Washington State waters, Ecology’s Water Quality Program made an interim policy decision to disallow the use of copper in salmon-bearing waters in May 1999. This decision affects all waters of the state utilized by native salmonids and will be revisited in the risk assessment exercise scheduled for copper compounds next year.

Irrigation Ditches Herbicides are used throughout Washington State on the canals, laterals, drains, and waterways of irrigation systems to maintain flow velocity and capacity of the waterways that drain into various streams and rivers, including the Columbia. Commonly used herbicides include 2,4-D, copper sulfate, acrolein, and xylene. Application practices vary somewhat but typically 2,4-D is applied to control terrestrial vegetation along canal and drain banks 1-2 times/year during the growing season. Irrigation districts usually apply copper sulfate to control filamentous green algae during the growing season. Copper sulfate may be applied every two weeks, generally to the laterals. Most districts use acrolein to control in-water vegetation (Weaver, 1999). Applications of herbicides are allowed by letter from Ecology to the irrigation districts but the districts have been encouraged to develop a separate EIS document for guidance regarding these applications.

4. Experimental Use Permits

Pesticides are allowed on an experimental use basis for purposes of research or in an emergency. Emergency situations occur every year in Washington State, taking an economic toll. Section 18 of FIFRA, a provision that allows EPA to temporarily exempt a pesticide from the full requirements of registration under emergency circumstances, specifically deals with these emergency situations. Because the state of Washington is one of the leading minor crop states in the nation and grows over 300 different commercial crops, we have a fair number of emergencies each year.

WSDA and Ecology have a concurrent process for issuing Experimental Use Permits for pesticides that are not federally registered for aquatic use. All aquatic pesticides are classified as “restricted-use” in Washington State and therefore require a license and aquatic endorsement for application. For pesticides currently not allowed by Ecology, but registered by WSDA, Ecology issues Experimental Water Quality Permits for large scale treatments, including whole lake treatments.

RCW 90.48.445 exempts small scale EUPs as defined in 40 CFR Section 172.3 from SEPA. When WSDA issues an experimental use permit (as authorized by RCW 15.58.405(3)), the exception from SEPA is limited to experiments of one surface acre or less (Substitute Senate Bill 5670, 1999). Experimental use for sites larger than one surface acre is still subject to SEPA review under federal law.

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Section II. Alternative Aquatic Plant Management Methods

A. Introduction to Alternatives

As stated in the Fact Sheet, the Proposed Action that triggered this SEPA action is the use of herbicides for aquatic plant control. This includes updating our information on the proposed use of new and currently allowed herbicides where significant research or information has been developed since our last assessment. 2,4-D, Aquathol and Hydrothol 191 are evaluated in year 2000, and trichlpyr, diquat and copper compounds will be evaluated in 2001. When the first Aquatic Plant Management EIS was developed in 1980, Ecology determined that the application of herbicides is likely to result in significant adverse environmental impacts, triggering the need to develop an Environmental Impact Statement (EIS). The evaluation of these chemicals adds information and analysis to supplement the 1980 and subsequent 1992 Aquatic Plant Management Environmental Impact Statements.

SEPA uses the EIS process to identify and analyze probable adverse environmental impacts, reasonable alternatives, and possible mitigation. The EIS process provides public participation in developing and analyzing information and improves the proposals through mitigation of identified adverse environmental impacts and development of reasonable alternatives that meet the objective of the proposal. It also gives agencies the authority to condition or deny a proposal based on the agency's adopted SEPA policies and environmental impacts identified in a SEPA document (RCW 43.21C.060, WAC 197-11-660).

This SEIS discusses five alternatives for controlling aquatic plants. Permit applications are usually submitted for herbicide applications for a one-year treatment, with Ecology receiving the same herbicide applications for the same water bodies year after year.

Along with the current evaluation of the proposed action, Ecology updated some of the information on the alternative methods for control of aquatic vegetation. This update has been largely cursory due to lack of funding and time to update the whole EIS. The preferred method outlines a planning component to encourage the use of integrated management methods in the permitting process as recommended by the 1997 Integrated Pest Management (IPM) Law (Chapter 17.15 RCW). The planning guidance also takes advantage of the 1997 changes to the WQS (WAC 173-201A-110) which enable Ecology to authorize a three to five year permit under certain conditions. ESA issues and the development of biological control methods are also changing the permitting process and these changes are further discussed in the Preferred Alternative Section.

Any aquatic plant control method may result in adverse environmental impacts. For this reason, the principle features and mitigation measures for each alternative are discussed in detail at the end of their respective sections. The information provided is intended to aid decision-makers in assessing available alternatives. Alternatives evaluated are:

1. Use of an integrated management approach (the preferred alternative),
2. The "no action" alternative, (continuing current practices),
3. Use of mechanical/manual methods only,
4. Use of biological methods only, and
5. Use of chemical methods only (the proposed actions).

Alternatives are defined in terms of actions that might be taken by an agency or agencies for aquatic plant management. The "action(s)" required to implement various aquatic plant management alternatives include state activities such as Ecology's issuance of short term modifications of water quality standards to allow rotoation, suction drudging or application of herbicides to waters of the state. Actions may also include Ecology's funding of lake restoration and freshwater aquatic plant management activities and WDFW issuance of permits allowing the use of grass carp and their issuance of HPAs for hand pulling, raking, harvesting diver dredging, weed rollers, rotoation and bottom barrier installation. Local governments may require shoreline permits for mechanical or chemical treatment projects or projects costing over \$2,500. The U.S. Army Corp of Engineers may also require Section 404 permits for suction dredging and rotoation projects. For simplicity, the term "permits" is used when referring collectively to all of these permits.

B. Criteria Used For Analysis and Comparison of Alternatives

State surface water quality regulations and standards (RCW 90.48; Chapter 173-201A WAC) provide authority to establish criteria for waters of the state and to regulate various activities, including those related to aquatic plant control. These standards articulate an intent to protect public health and maintain the beneficial uses of surface waters, including: recreational activities such as swimming, SCUBA diving, water skiing, boating and fishing and aesthetic enjoyment; public water supply; stock watering; fish and shellfish rearing, spawning, and harvesting; wildlife habitat, and commerce and navigation. A short-term modification of water quality standards (permit) cannot be issued if water quality degradation interferes with or becomes injurious to existing water uses and causes long-term harm to the environment.

Key to the analysis and comparison of alternatives is the state's goal to maintain beneficial uses of state waters and protect the environment. Therefore each method will be evaluated for:

1. The extent the alternative detracts from the beneficial use of a particular water body;
2. Potential adverse environmental impacts, especially regarding ESA listed species and wetlands;
3. Potential adverse human health impacts; and
4. The degree to which any one method effectively controls a particular plant problem, especially those aquatic plants designated as noxious or invasive.

Because of the complexity and variability of water bodies, their beneficial uses and the types of management needed, specific evaluation of impacts and mitigation will have to be applied on a case-by-case basis to various management proposals. To assist in this assessment, each method and each herbicide allowed for use will be assessed with the above criteria. If adverse environmental impacts cannot be avoided by the use of any one method or herbicide, its use may be severely restricted or disallowed.

In the sections on various methods of aquatic plant management, and for each herbicide assessed by this Supplemental Environmental Impact Statement, elements of the environment (WAC 197-11-444) that may be significantly affected are discussed. Since lakes are the primary environment where methods of aquatic plant control will be applied, only those elements that pertain to lakes, ponds or streams and their beneficial use are included in the assessment.

C. Mitigation Defined

As defined by SEPA, mitigation means, in the following order of preference:

1. Avoiding the impact altogether by not taking a certain action or part of an action;
2. Minimizing impacts by limiting the degree or magnitude of the action and its implementation by using appropriate technology, or by taking affirmative steps to avoid or reduce impacts;
3. Rectifying the impact by repairing, rehabilitating, or restoring the affected environment;
4. Reducing or eliminating the impact over time by preservation and maintenance operations during the life of an action; and
5. Compensation for the impact by replacing, enhancing, or providing substitute resources or environments.

When evaluating potential impacts to habitat, the following definition shall be used: wildlife habitat means waters of the state used by, or that directly or indirectly provide food support to fish, other aquatic life and wildlife for any life history stage or activity.

D. ESA Considerations for all Methods

Several salmon populations and other aquatic biota are listed for special protection under ESA. Such listings may affect aquatic control projects all over Washington State. Information regarding potential listings of endangered species in particular water bodies can be obtained from the local office of WDFW.

Obtaining a permit from Ecology for the application of herbicides does not exempt an applicator from “Take” liability under ESA. Applications that are made outside the permitting process, such as the 2,4-D applications being made under SSB 5424, or in irrigation ditches are also not exempt from potential take liability. “Take” means to “harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in such conduct” with respect to a species listed under ESA (16 U.S.C. Section 1532(19)). Current permit applications require applicators to state whether the proposed treatment area is part of any designated critical habitat of an ESA listed species or an Evolutionary Significant Unit listed under ESA. Proposed treatments that may have an adverse impact on a listed species may be denied a permit or restricted.

E. Wetlands: Mitigation for All Methods

Definitions. Evaluation of potential adverse impacts to wetlands from aquatic plant control will be determined using the following definitions.

1. "Wetlands" means those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions, such as swamps, marshes, bogs, and similar areas. This includes wetlands created, restored or enhanced as part of a mitigation procedure. This does not include constructed wetlands or the following surface waters of the state intentionally constructed from non-wetland sites: Irrigation and drainage ditches, grass-lined swales, canals, agricultural detention facilities, farm ponds, and landscape amenities.
2. "Constructed wetlands" means those wetlands intentionally constructed on non-wetland sites for the sole purpose of wastewater or storm water treatment and managed as such. Constructed wetlands are normally considered as part of the collection and treatment system.
3. "Created wetlands," means those wetlands intentionally created from non-wetland sites to produce or replace natural wetland habitat.

4. "Drainage ditch," means that portion of a designed and constructed conveyance system that serves the purpose of transporting surplus water.
5. "Irrigation ditch means that portion of a designed and constructed conveyance facility that serves the purpose of transporting irrigation water from its supply source to its place of use.

The following provides guidance for decisions regarding wetlands mitigation:

".... The overall goal of mitigation shall be no net loss of wetland function and acreage. Where practicable, improvement of wetland quality should be encouraged."

1. Water quality in exceptional wetlands shall be maintained and protected. Exceptional wetlands are those determined by Ecology to meet one of the following criteria:
 - Wetlands that are determined by the Department of Natural Resources to meet the criteria of the Washington Natural Heritage Program as specified in Chapter 79.70 RCW:
 - Mapped occurrence of threatened and endangered species and their priority habitats as determined by WDFW:
 - Documented critical habitat for threatened or endangered species of native anadromous fish populations as determined by WDFW:
 - Designated outstanding resource waters. and
 - High quality, regionally rare wetland communities with irreplaceable ecological functions, including sphagnum bogs and fens, marl fens, estuarine wetlands and mature forested swamps.
2. Water quality in all other wetlands shall be maintained and protected unless it can be shown that the impact is unavoidable and necessary. Avoidance shall be the primary means to achieve the water quality goals of this chapter. For water-dependent activities, unavoidable and necessary water quality impacts can be demonstrated where there are no practicable alternatives that would:
 - Not involve a wetland or that would have less adverse water quality impacts on a wetland;
 - Not have other more significant adverse consequences to the environment or human health.
3. When it has been determined that lowering the water quality of a wetland is unavoidable and necessary and has been minimized to the maximum extent practicable, wetland losses and degradation shall be offset, where appropriate and practicable, through deliberate restoration, creation, or enhancement of wetlands.
 - In-kind replacement of functional values shall be provided, unless it is found that in-kind replacement is not feasible or practical due to the characteristics of the existing wetland and a greater benefit can be demonstrated by an alternative. In such cases, substitute resources of equal or greater ecological value shall be provided.
 - On-site replacement shall be provided, unless it is found that on-site replacement is not feasible or practical due to physical features of the property or a greater benefit can be demonstrated by using an alternative site. In such cases, replacement shall occur within the same watershed and proximity.
 - A mitigation plan shall be required for proposed mitigation projects. Elements that may be required in a mitigation plan include:
 - a. A description of the impact or damage that is being mitigated.
 - b. A description of the mitigation site,
 - c. A discussion of the goals of the mitigation, e.g., restoring a native plant community, enhancing the wildlife habitat values by diversifying vegetation, replacing native aquatic vertebrates, etc.
 - d. A description of actions being taken, e.g., planting, habitat enhancement, restocking, etc., and
 - e. A monitoring plan to determine if the actions achieve the goals.
 - Restoration, enhancement, or replacement shall be completed prior to wetland degradation, where possible. In all other cases, restoration, enhancement, or replacement shall be completed prior to use or occupancy of the activity or development, or immediately after activities that will temporarily disturb wetlands.

Section III. The Preferred Alternative: An Integrated Aquatic Plant Management Plan

A. Documents and References for Developing an Integrated Aquatic Plant Management Plan

The current preferred alternative is based on the 1992 Final Supplemental Environmental Impact Statement (FSEIS) and includes new guidance from:

- *A Citizen's Manual for Developing Integrated Aquatic Vegetation Management Plans* (1994),
- The 1997 Integrated Pest Management (IPM) Law (Chapter 17.15 RCW), and
- The 1997 changes to the WQS (WAC 173-201A-110).

Integrated aquatic plant management planning has already been implemented with some success. At least a dozen plans have been written to address various nuisance or noxious weed problems in lakes in Washington. Ecology continues to actively urge lake groups that chemically treat their lakes regularly to develop an integrated aquatic plant management plan before they apply for future chemical/aquatic plant control permits. The level of planning needed may be based on the size or percentage of the waterbody to be treated; however, Ecology recognizes there is no one-size-fits-all planning method and recommends an appropriate level of planning be used when applying for chemical/aquatic plant control permits.

- Watershed planning is the broadest, most inclusive planning method and is probably most appropriate for use by governmental entities and other large groups able to secure grants or other funding for the plan.
- Lake Management planning is a somewhat reduced scale of watershed planning but still contains some critical components of the larger plan. Typically lake management groups and other, small-scale groups may consider this level of planning for aquatic plant control.
- Lastly, individuals or small groups with limited resources may consider integrated aquatic plant management planning on a scale that fits their needs. This last type of planning would still incorporate critical components of the other two methods but would be doable for small-scale management operations.

Like the 1992 preferred alternative, Ecology's 1994 guidance manual and the IPM law require consideration of all available methods in an integrated aquatic plant management plan. Under this alternative, each lake or surface water system is evaluated to determine the extent and underlying causes of aquatic plant and/or algae problems and the most effective and environmentally sound control strategy for correction and long-term management. Using the best combination of biological, mechanical, and physical control methods may eliminate the need for further action against many nuisance aquatic plants. When the nuisance plant species can not be controlled with non-chemical methods at a level adequate to support the prioritized beneficial uses, the addition of chemical control methods to the management strategy may be necessary or desirable, especially when targeting noxious species. This current supplement to the EIS looks at selected chemicals as additional tools for aquatic plant management. However, when chemicals are added to a management strategy, the selection of the herbicide, dosage, and treatment time must be carefully coordinated to avoid ecological disruptions.

In general, integrated management is the selection, integration, and implementation of control methods based on predicted economic, ecological, and sociological consequences. This concept is based on the premise that, in many cases, no single control method will by itself be totally successful. Thus, a variety of biological, physical, and chemical control and habitat modification techniques are integrated into a cohesive plan developed to provide long-term vegetation control (Bottrell 1979). Integrated management may include various land-use practices necessary to reduce or eliminate the introduction of nutrients and sediments causing accelerated aquatic plant and/or algae growth. The ultimate objective is to control detrimental vegetation in an economically efficient and environmentally sound manner.

The IPM Law, Chapter 17.15 RCW, defines the elements of integrated pest management to include:

- (a) Preventing pest problems,
 - (b) Monitoring for the presence of pests and pest damage,
 - (c) Establishing the density of the pest population, that may be set at zero, that can be tolerated or correlated with a damage level sufficient to warrant treatment of the problem based on health, public safety, economic, or aesthetic thresholds;
 - (d) Treating pest problems to reduce populations below those levels established by damage thresholds using strategies that may include biological, cultural, mechanical, and chemical control methods and that must consider human health, ecological impact, feasibility, and cost-effectiveness; and
 - (e) Evaluating the effects and efficacy of pest treatments.
- (2) "Pest" means, but is not limited to, any insect, rodent, nematode, snail, slug, weed, and any form of plant or animal life or virus, except virus, bacteria, or other microorganisms on or in a living person or other animal or in or on processed food or beverages or pharmaceuticals, which is normally considered to be a pest, or which the director of the department of agriculture may declare to be a pest.

Typically, this approach would not be used for one-season treatments but would rather be the basis for three to five year aquatic plant treatment plans. A key use of a plan would be its development and assimilation into the permit process as provided by WAC 173-201A-110 (1)(c). Ideally, the permit would provide guidance and consistency for balancing various beneficial uses and control methods for each aquatic system. Each permit would be developed through a public involvement process consistent with SEPA and the Administrative Procedure Act (Chapter 34.05 RCW) that includes state and local resource agencies, Indian tribes, user groups and the public. Proposed integrated management planning should be set up so that affected communities and interest groups can review and comment on proposed management strategies where potentially conflicting uses in a given water body exists. Plans would be used to help lake managers and permit writers evaluate whether plants that provide fisheries or wildlife habitat should be eradicated to improve aesthetics or recreational use of a waterbody. Resource agencies would be asked to participate in plan development and review. These agencies, including Ecology, would have to ensure consistency of plans with agency goals, policies, and regulations and each plan would be subject to Ecology's review and approval before use in the permitting process.

B. Guidelines for Developing an Integrated Aquatic Plant Management Plan

An illustrated manual entitled *A Citizen's Manual for Developing Integrated Aquatic Vegetation Management Plans* (IAVMP Manual) is available for the development of a watershed, lake or an integrated aquatic vegetation management plan (Gibbons, 1994). The IAVMP Manual, dated January 1994, was written to assist citizens and lake management groups to develop IPM plans. The manual (about 40 pages not including the appendices) is available on Ecology's website at: <http://www.wa.gov/ecology/wq/plants/management/manual/index.html> or a copy may be obtained from Ecology's publications office at (360) 407-7472. A sample integrated aquatic vegetation management plan developed for Lake Leland is also available on Ecology's website at: <http://www.wa.gov/ecology/wq/plants/leland/index.html>. The IAVMP Manual and the sample plan specifically address controlling nuisance aquatic plants and provide guidance for aquatic plant managers. Aquatic plant managers are those individuals and entities interested in or responsible for sponsoring and/or providing oversight for aquatic treatments designed to control nuisance aquatic pests. Funding may be available for the development of integrated aquatic vegetation plans through Ecology's Aquatic Weeds Program. Funding is for government entities, tribes or special purpose districts for use on waterbodies with public boat ramps. Noxious weed projects receive funding priority. For more information see <http://www.wa.gov/ecology/wq/plants/grants/focusgrant.html>. The following is a summary of the IAVMP Manual guidance.

Identify the aquatic plant or pest targeted for control

The first step in preparing a plan is the development of a problem statement. The problem statement considers the users of the waterbody and what they consider to be the problem. Those problems can be grouped into categories and condensed into a problem statement.

Aquatic plant communities vary at least as much as the human and wildlife communities that use them, necessitating the consideration of many factors for potential aquatic plant managers, such as:

- Is there an aquatic plant problem?
- What is the problem?
- Should anything be done about it?
- Should a community group be formed to address the problems?
- Who will participate in the planning process?

Depending on a water body's size, depth, and other characteristics, aquatic plant growth can be extensive or occur in small localized areas. In order to design an effective management program specific to the water body, the types of aquatic plants growing there, their location and the extent of growth must first be determined. This can be accomplished by performing an aquatic plant survey. A survey involves systematically traveling around the water body and shoreline and noting aquatic plant conditions. An important part of the survey is collecting samples of aquatic plants to verify the species. This is especially important if invasive, nonnative aquatic plants are suspected to be present.

Once the aquatic plants are mapped, the next step is to use that information to write a description of beneficial and problem plant zones. Characterizing the aquatic plant zones helps to determine where special control actions are required and consists of the following tasks:

1. Describe Plant Types
2. Determine Problem Areas and Beneficial Plant Zones
3. Determine Need for Special Action

Control and/or eradication of aquatic species listed as noxious is considered more critical than control of non-noxious species. The Washington State Noxious Weed Board designates certain aquatic plants as noxious. None of the weeds on the Washington State Noxious Weed List are native to the state. Every year, the Board adopts, by rule, a noxious weed list. The list determines which plants will be considered noxious weeds and where in Washington control will be required. Noxious weeds are divided into classes (Class A, B, or C), depending for the most part on the extent of distribution within Washington.

- Class A species are those noxious weeds not native to the state that are of limited distribution or are unrecorded in the state and that pose a serious threat to the state,
- Class B species consists of those noxious weeds not native to the state that are of limited distribution or are unrecorded in a region of the state and that pose a serious threat to that region,
- Class C species have populated the state to such an extent that containment may not be practical.

This approach classifies non-native plants that have the potential to cause serious problems because they are invasive and/or are a threat to natural resources such as native-plant communities, wetlands, rangeland, or cropland. An integrated aquatic plant management approach recognizes the need for a strategy of total eradication under special circumstances. In some cases, impacts and potential impacts from noxious or invasive non-native species may outweigh impacts and potential impacts from treatment.

Requirements for control are region-specific and based on the economic and environmental feasibility for effective control along with the seriousness of problems presented by the noxious species. The fact that control is required and enforced should be considered an indication of the feasibility of control in addition to the seriousness of the problem presented by a noxious weed. Noxious plant species that have been identified are on the State Noxious Weed List (Chapter 16-750 WAC and can be found at http://www.wa.gov/agr/weedboard/weed_laws/wac.html).

Public Involvement and Education

Once an aquatic-plant growth problem has been recognized, it is crucial to bring all interested and affected parties together early on to participate in planning. Identifying people who have an interest in the water body often requires a bit of searching. The water body may serve a variety of groups with sometimes-conflicting interests. State, county or local governments and agencies may be involved. Private businesses or other interest groups may have concerns about the water body as well. Some groups that may have an interest in management of an aquatic system are:

- Residents or property owners around the water body
- Special user groups (e.g., bass anglers, Ducks Unlimited)
- Local government
- State and federal agencies (e.g., State Department of Ecology)
- Native American tribes
- Water-related businesses (e.g., resorts, tackle & bait shops, dive shops)
- Elected officials
- Environmental groups (e.g., Audubon).

Certainly every effort should be made to bring as many interested parties to the table as possible. However, it may be difficult and costly for an individual shoreline owner or other small groups interested in aquatic

plant management to identify and contact potentially interested groups, conduct public meetings and keep the community informed. Conceivably, potential aquatic pest managers may elect to have their plans developed in conjunction with their permits for this reason.

As explained in Appendix A, applications for short-term water quality modifications are now forwarded for SEPA review and comment to WSDA, WDFW, DNR, tribes, local governments, other Ecology offices and programs, and interest groups, initiating a ten to twenty-one day comment period. The Administrative Procedure Act additionally requires public notice of the permit application and the draft permit to give all interested parties an opportunity to comment on the proposed permit. Public notices of application and of draft permits may be done on a batch basis if desired. Public hearings are required when a permit section manager deems one appropriate to inform the public or if there is sufficient interest and a likelihood of meaningful public comment to warrant one. Open-house (informal, informational) meetings should be offered if needed. Comments received are included in the official permit record, and Ecology prepares a response to comments explaining its acceptance of the permit, changes that were made between the draft and final permit and the reasons for those changes.

Once a problem statement has been drafted and all interested and affected parties have been invited to participate in the planning effort, the next step is to come up with specific management goals. Management goals define what the community wants to achieve in response to the aquatic plant problems. Defining goals helps in selecting the best methods that form the heart of the plan.

State a Management Objective in Support of Beneficial Uses

Beneficial uses of water bodies are protected by Washington State statute and must be compatible with its capacity to sustain those uses, both human and natural. Unfortunately, a single water body often supports many different desirable uses, which sometimes conflict with each other. The management challenge involves identifying and agreeing on uses that complement each other, and realistically managing for these uses.

Lakes are eco-systems that provide habitat for fish, wildlife and aquatic plants. The plan to control aquatic plants and algae should consider what the lake would naturally support in a pre-development stage. Then a decision should be made on how much control is desired. Should the algae and plant populations be:

- Kept the same as present conditions,
- Returned to a 'natural' pre-development condition, if possible,
- Controlled beyond what the lake would naturally support and to what extent?

Under the alternatives to restore or control beyond restored conditions, each lake system is evaluated to determine the extent and underlying causes of aquatic plant problems. Then, the most effective and environmentally sound control strategy can be implemented. The following points should be considered in developing a management objective.

1. The ecosystem is the management unit and the entire watershed should be managed as a natural ecosystem or if needed, restored to a natural system. Even subtle manipulations may affect the ecosystem, possibly aggravating one problem in attempt to resolve another. System disruptions should be avoided, and problem vegetation held to a tolerable level. However, the goal for species designated as noxious would be total eradication, maintenance at low levels, or containment.

2. Any technique, or combination of techniques, must be carefully considered in an ecological context before and after use of aquatic plant or algae control. As demonstrated in the impact analysis sections of this SEIS, most alternatives have the potential to cause some level of adverse environmental impacts.
3. Integrated management requires review of each waterbody using an interdisciplinary approach. When determining if there is an aquatic plant/algae problem and before deciding how to solve a particular plant-management problem, the waterbody should be evaluated from several perspectives. This may require identification of the cause of suspected excessive plant and/or algae growth including: sources of nutrient loading, an analysis of water and sediment quality, an assessment of beneficial uses provided by the water body, and identification of any wetlands or other sensitive ecosystem in the area. Proposals should be reviewed by a variety of experts or agencies that specialize in different fields of lake management. Special interest groups and waterbody users would also be involved in this evaluation.
4. A "risk" threshold should be established to help determine if plants proposed for eradication are truly problematic. Though dozens of plant species may exist in a given waterbody, only a few may present major problems in any one location. The threshold would be used to determine if, and the degree to which, an aquatic plant should be controlled, contained, or maintained at low levels. (Also see Chapter 17.15 RCW (c), of the IPM law.)

Ecology as well as private contractors provide information about waterbody management planning or other aspects of aquatic plant management. This includes lectures or participation in conferences designed for herbicide applicators, lake management associations and districts, weed control boards, resource agencies, academicians or others that may be interested in, or affected by, aquatic plant management efforts. WebPages on aquatic plants and lake issues are maintained by Ecology at <http://www.wa.gov/ecology/wq/links/plants.html>. Information about management methods, noxious weeds, native plants, plant identification, financial assistance for weed management projects, and general information about lakes is available at this site. Publications about noxious aquatic weeds are also available and from Ecology's Publication Office at (360) 407- 7472.

After management objectives for the water body are determined, the physical characteristics of the water body must be assessed for prevention and restoration opportunities.

Prevention and Lake Restoration Opportunities

A lake or river is a dynamic, living system, teeming with physical, chemical and biological activity. The system extends beyond its shores to include surrounding land whose waters drain into the water body (the watershed). A water body and its watershed are inseparable. In fact, water body conditions are very much influenced by what occurs in the watershed. For instance, a watershed contributes nutrients to a water body that are necessary for aquatic plant growth. These nutrients—especially phosphorus and nitrogen—flow to the lake from all parts of the watershed by way of streams, ground water, and stormwater runoff. In addition, activities in the watershed, such as agriculture and forestry, road maintenance and construction can all contribute silt, debris, chemicals, and other pollutants to the waterbody.

A plan should consider these possible sources of nutrient inputs and identify long-term measures to reduce them. Controlling watershed inputs from these sources can potentially enhance the effectiveness of primary in-lake control measures. Therefore this planning step is composed of developing a map that:

1. Describes the watershed, including characteristics such as:
 - Size and boundaries of the watershed
 - Tributaries, wetlands and sensitive areas
 - Land use activities in the watershed
 - Nonpoint pollutant sources
 - Existing watershed management, monitoring or enhancement programs
 - The presence of rare, endangered or sensitive animals and plants

2. Describes the waterbody. Waterbody features that are important to identify are:
 - Location
 - Size, shape, and depth
 - Water sources
 - Physical and chemical characteristics (water quality)
 - Biological characteristics (animals and plants)
 - Shoreline uses
 - Outlet control and water rights.

3. Identifies beneficial use areas such as:
 - Conservancy areas, including habitats that are integral to the lake ecosystem or wildlife, such as nesting sites, fish rearing or spawning areas, or locations of rare plant communities,
 - Boating and boat access areas (launches, ramps),
 - Water skiing zones,
 - Beaches and swimming areas (public, private),
 - Fishing areas,
 - Areas for special aquatic events (e.g., sailing, rowing, mini hydroplane races),
 - Parks, picnic areas, nature trails, scenic overlooks,
 - Irrigation/water supply intakes, and
 - Other shoreline uses (e.g., residential, commercial).

For small treatment proposals, watershed-mapping requirements may not be necessary. However, much of this information is readily available in county Growth Management Act (GMA) or other planning documents, maps or data that can be obtained from local planning or public works departments and state agencies. Check with the local WDFW office for ESA species of concern.

Preventing algae and aquatic plant problems includes preventing the introduction of noxious species, promoting eradication of noxious species to keep them from spreading to new areas, and improving water quality. The first goal, preventing introduction of noxious species, is achieved through efforts by Agriculture's quarantine program, Ecology's freshwater aquatic weeds program, developing a state Aquatic Nuisance Species Plan or developing some level of an integrated aquatic plant management plan. Eradication of some noxious species from a waterbody may be possible using a combination of aquatic plant control methods, and is further discussed in Ecology's "Washington's Water Quality Management Plan to Control Nonpoint Source Pollution" (2000). An overview of prevention techniques available for improving water quality is also summarized in the Nonpoint Source Pollution Plan. The Plan describes a holistic approach to controlling and cleaning up nonpoint source pollution, including lake restoration activities, which may be appropriate for large-scale watershed planning activities.

After management objectives for a waterbody are determined and the physical characteristics of the water body are known, control methods can be determined for a management plan.

Identify Control Methods

At this time, choices available for aquatic plant control include manual and mechanical methods, biological methods, and chemical methods. All are reviewed in this document or are discussed in the IVAMP Manual and on Ecology WebPages. As discussed above, a decision to use one or more methods is based on potential environmental impacts, available mitigation, the amount and type of vegetation to be removed and efficacy of control and cost. In most cases, achieving control of aquatic plants without use of herbicides is preferred, particularly where target populations are small and manual methods or bottom barriers are a practical alternative.

Management strategies may involve a mix of methods. For example, for some waterbodies it may be best in the long term to develop a Eurasian watermilfoil strategy designed to eradicate rather than control the species. The goal of eradication would be to eliminate the species from a system and may require measures more extreme than would be required for control. However, all large-scale control strategies that require repeat treatments may, over time, result in impacts that exceed those associated with eradication. An eradication program may include mechanical harvesting to reduce biomass, treatment with herbicides to achieve eradication, and if required, follow-up “spot” treatments that may include a combination of methods, including hand pulling, diver dredging, or spot application of aquatic herbicides.

Control intensity also needs to be specified. Are there plant zones around the lake that should be left alone (**no control**)? Where should a **low level of control** be applied to preserve some intermediate level of plant growth? And under what circumstances would a **high level of control** be necessary, such as where a minimal amount of nuisance plants can be tolerated (i.e. public swimming beaches).

And finally, a plan for monitoring the effectiveness and impacts of various control methods at selected sites on selected species must be incorporated into the integrated treatment plan. Before and after pictures as well as water samples and plant surveys are ideal tools for assessing the effectiveness of the chosen integrated treatment plan.

Choosing an integrated treatment scenario

This step involves choosing the combination of control efforts that best meets the needs of waterbody users with the least impacts to the environment. The procedure consists of evaluating each control option available using an **integrated vegetation management approach**. This approach involves examining the alternatives with regard to such factors as:

- The extent of problem plant(s) infestation
- Scale, intensity, and timing of treatment effectiveness against target plant(s),
- Duration of control (short-term vs. long-term)
- Human health concerns (if any)
- Environmental impacts and mitigation, if needed
- Program costs
- Permit requirements (federal, state, local).

Reviewing control alternatives in light of these and other site-specific factors provides a means of narrowing the options into an appropriate management package. This SEIS contains information on the impacts and mitigation requirements for each proposed method and those sections which describe the

chosen methods should be carefully considered. No management program, however, is without some impacts. Choosing a management program will require weighing all the factors. The trick in deciding a course of action is to achieve a **balance** between expected management goals at a reasonable cost and acceptable environmental disruption.

Further discussion of how to develop an integrated aquatic plant management is provided in the IAVMP Manual. Once a plan is developed it may be included in an application for a Short-term Water Quality Modification (Permit) and submitted to Ecology for processing. If an Ecology permit will not be needed to implement the actions in the plan, the final task is to take all the information and formulate a **long-term action program (plan)** for aquatic plant management. This Plan provides the community with guidance and direction for aquatic plant management. The decision to proceed with aquatic plant control in the waterbody is just the beginning. Follow-through is critical. **Aquatic plant control is an ongoing concern that requires long-term commitment.** This is particularly true of water bodies with exotic plants or with nuisance plant growth that has developed over many years. In these situations, achieving management goals could take many years. The Plan should be flexible and evolving. It should provide for regular checking of how well the actions are working and allow for modification as conditions change.

C. Impacts and Mitigation

The impact of aquatic plant control methods selected for use, including the impact of removal of targeted species, must be assessed in terms of impacts on the particular ecosystem. This is a significant requirement in that the manipulation of an ecosystem may aggravate some pest problems while managing other pest populations. As demonstrated in the impact analysis section of this SEIS, most alternatives have the potential to cause some level of adverse environmental impacts. Even subtle manipulations may affect the ecosystem, possibly aggravating one problem in attempt to resolve another. Integrated management manipulates ecosystems to hold nuisance vegetation to tolerable levels while avoiding disruptions of the systems (Smith and van den Bosch 1967). Thus, all proposed techniques, or combination of techniques, must be carefully considered in an ecological context before and after use of aquatic plant controls. To do this a plan for monitoring the effectiveness and impacts of various control methods at selected sites on selected species must be developed. And finally, each alternative section contains mitigation measures that may apply. These measures should be included in the final plan, and the monitoring requirements as well as whatever mitigation measures are needed will be incorporated, when appropriate, into the conditions of the permit or the final action plan.

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Section IV. Alternative 2: No Action: Continuing Current Practices

A. Description of the No Action Alternative

The no action alternative means that Ecology would continue to issue water quality modifications and grant funds for aquatic plant control as we have since 1992. Ecology would continue to participate in lake restoration activities such as aeration, dilution, lake level regulation, and watershed controls and continue funding freshwater aquatic plant management activities through the Aquatic Weed Management Fund.

If new or “improved” aquatic vegetation control herbicides are not assessed and subsequently not permitted, opportunities to have new herbicide formulations for aquatic plant control that may be less harmful to the environment and humans and that are less costly will not be available to Washington State citizens. This may result in more exotic and invasive plant infestations in Washington waters

The Washington Legislature directed Ecology to expand certain chemical application sections of the 1992 SEIS to make it more responsive for the application of new, commercially available herbicides, and to evaluate their use with the most recent research available (Engrossed Substitute Senate Bill 5424, effective May 10, 1999). If Ecology simply continued current practices, it could find itself at odds with a legislative directive.

If current practices are maintained, all activities related to plant management will not be pulled together in an integrated approach for long-term vegetation control. This could result in inefficient, ineffective, and/or environmentally unsound control practices. Without a comprehensive and coordinated review of each waterbody for which chemical treatment is proposed, the problems causing excessive plant control may be allowed to continue and control options may not be thoroughly examined. Repeat treatments and associated impacts may occur if underlying causes continue to create an environment in which nuisance aquatic plants and algae thrive. As long as there are nuisance plants, impacts associated with control and eradication activities would continue.

B. Potential Impacts and Mitigation under Continuing Current Practices

Currently, short-term modifications of water quality standards (permits) are processed for certain federal and state registered aquatic herbicides. In 1999, Ecology received water quality modification applications for the following herbicides:

- copper compounds (including Komeen, Copper Sulfate, and AV-70);
- glyphosate (Rodeo),
- fluridone (Sonar),
- endothall (Aquathol K),
- Hydrothal 191 – experimental use,
- diquat dibromide (diquat) – experimental use,
- 2,4-D ester (Aqua-Kleen) – legislative allowance
- triclopyr (Renovate) – experimental use.

Before issuing permits, proposals are evaluated relative to their impact on human health, unique ecosystems, potable and irrigation water supply, fish, wildlife, navigation, hydropower, and other beneficial uses of state waters. Permits issued contain conditions designed to protect the environment and human health. Categories of conditions include, but are not limited to:

- Buffers, including restrictions on timing, distance, and chemical application rates,
- Notification requirements,
- Regulatory compliance, including compliance with the herbicide label and all applicable local, state, and federal regulations,
- Application methods,
- Monitoring, and
- Compensatory mitigation.

The current herbicide application review process allows for review of the application and associated environmental documents by state agencies, Indian tribes, local agencies, and the public. Comments or concerns received during the review process are carefully considered and integrated into permit conditions where appropriate. This process allows for coordination of actions related to issuance of water quality modifications for aquatic herbicide applications. However, other activities related to aquatic plant management, such as mechanical harvesting, installation of bottom barriers, weed rolling, funding lake restoration activities and watermilfoil control, and issuing permits for rotoation or introduction of grass carp are not coordinated through this process. Under the current system, isolated actions related to aquatic plant management may be taken by a variety of divisions within one or more agencies, funded through separate mechanisms, and carried out under independent mandates.

The 1992 SEIS recommends an integrated approach to aquatic plant management and allows the use of copper, endothall, fluridone and glyphosate to control various types of aquatic plants. The integrated pest management approach identified in the 1992 SEIS as the preferred alternative for controlling nuisance aquatic plant populations allows for the use of the most efficient and effective control method, or combination of control methods, while minimizing impacts to human or environmental health. It was found that having a variety of control methods available provides the flexibility necessary to control nuisance populations of native as well as invasive non-native species in situations where it is desirable to maintain other, often conflicting beneficial water uses. Having the most up-to-date aquatic herbicides is equally important to most efficient and effective control method, or combination of control methods, for use for aquatic plant control. Where no change is made to the existing program, and no consideration of new products is provided, efficiencies may be compromised. The 1999 Washington Legislature also directed Ecology to keep the aquatic plant management EIS current with new commercially available products so that control efforts can be as effective as possible. It is Ecology's intention to do so.

Section V. Alternative 4: Use of Mechanical/Manual Methods

A. Introduction

Manual methods include hand pulling, cutting, and raking; mechanical methods include mechanical harvesting and cutting, weed rolling and rotoation. Bottom barriers and suction dredging are also included in this alternative.

Impacts associated with the exclusive use of mechanical and physical methods are usually short-term and relatively localized. Currently, many agency aquatic plant control programs process permits required for mechanical control, including general and individual Hydraulic Project Approvals (HPA) from Washington State Department of Fish and Wildlife (WDFW), shoreline permits from local agencies, Section 404 permits from the U.S. Army Corps of Engineers for diver dredging and rotoation and water quality modifications from Ecology. Under this alternative, Ecology would continue to administer funds for water quality improvement and aquatic plant control. Manual methods are generally more practical for small areas, such as those around docks, in swimming areas, and in areas containing obstructions. These methods are labor intensive but do not require substantial skill, equipment, or expense, and do not result in long-term adverse environmental impacts.

Environmental impacts associated with manual methods are expected to be minimal, however manual harvesting may result in short-term sediment disturbances with potential adverse impacts to water quality and associated biota, including threatened or endangered species if these species are not identified and avoided. When the use of manual methods is confined to small areas, it is expected that impacts would be short term and limited. However, harvesting and rotoation are generally performed on a larger scale and have the potential for wider scale impacts.

B. Bottom Barriers

Bottom barriers can be an efficient method for controlling small areas of problem aquatic plant populations, providing immediate removal from the water column and long-term control. Effectiveness varies depending on the type of barrier used, and control may range from 1-2 years up to 10 years or longer, as long as bottom barrier maintenance is regularly performed. Bottom barriers provide an attractive alternative to other types of control because they can be deployed and left in place for several growing seasons, eliminating the need for repetitive treatments.

Bottom barriers may interfere with fish spawning and may cause a significant decrease in the benthic community, but impacts appear to be limited to the treatment area. Bottom barriers are not selective within the treatment area, but when placed correctly, can be very selective for small, isolated areas. Wetland or "unique" species within the target area could be impacted unless they are identified and avoided.

1. Description

Covering sediment to prevent growth of nuisance aquatic plants is a management option employed since the late 1960s (Born et al., 1973, Nichols, 1974). A bottom barrier covers sediment like a blanket, compressing aquatic plants while reducing or blocking light. Once anchored to the sediment the barrier

compresses plant material into contact with microbially active sediments. Bottom barriers should be installed before aquatic plants have started growth in spring or, if installed later in the year, plants should be cut prior to the bottom barrier being placed. Materials such as burlap, plastic, perforated black Mylar, and woven synthetics can all be used as bottom screens. There are also commercial bottom screens that are specifically designed for aquatic plant control. These include:

- Texel® A heavy, felt-like, polyester material, and
- Aquascreen® A polyvinylchloride-coated fiberglass mesh which looks similar to a window screen.

An ideal bottom screen should be durable, heavier than water, reduce or block light, prevent plants from growing into and under the fabric, easy to install and maintain, and readily allow gases produced by rotting weeds to escape without "ballooning" the fabric upwards. Even the most porous materials, such as window screen, will billow due to gas buildup. Therefore, it is very important to anchor the bottom barrier securely to the bottom. Unsecured screens can create navigation hazards and are dangerous to swimmers. Anchors must be effective in keeping the material down and must be regularly checked. Natural materials such as rocks or sandbags are preferred as anchors.

Bottom barriers can provide immediate removal of nuisance plants and maintain a long-term plant-free water column. However, efficacy, durability, longevity, and cost of materials vary. Bottom barrier materials include polyethylene, polypropylene, synthetic rubber, burlap, fiberglass screens, woven polyester, and nylon film. The duration of control provided by a bottom barrier depends on several variables: the amount of fragment accumulation in the site originating from untreated areas, the rate of sedimentation (accumulated sediment may provide substrate for plant fragments to root), the degree to which plants can penetrate the barrier from the underside, and durability of the bottom barrier fabric. For example, burlap rots within two to three years, and plants can grow through window screening material. Regular maintenance can extend the life of most bottom barriers.

Bottom barriers are also one of the most expensive methods for aquatic vegetation control if used in a large-scale application. They are cost effective when used in small areas. Because the material and installation costs can be expensive, bottom barriers are generally applied to small areas such as around docks and in swimming areas. Texel® (needle punched polyester fabric) has been recommended for situations where routine maintenance can be performed and long-term control is desired. Burlap is suggested for low-cost, short-term (1 to 2 years) control. Burlap is recommended for early infestation projects where pioneering colonies of invasive exotic plants such as Eurasian watermilfoil are covered with this fabric which is then weighted with rocks or sandbags. In this instance, burlap is used to kill pioneering colonies. Burlap decomposes naturally allowing native species to colonize areas once occupied by invasive plants. Snohomish County personnel reported native species colonizing burlap bottom barriers that were placed over Eurasian watermilfoil plants in Lake Goodwin (Williams, 2000). He also noted that in colder waters, burlap remains intact longer than two years.

2. Impacts due to Bottom Barriers

Earth

Sediments Anchoring of bottom barriers may be difficult in deep soft sediments; thus their use in soft sediments may not be appropriate (Gibbons 1986). Additionally, removal of plants from the water column may affect the rate of sedimentation in the treatment area. Decomposing plants may increase sediment and barriers should be removed before they breakdown, unless they are specifically designed to do so.

A specific concern is the limitation of barrier performance resulting from sediment gas evolution following placement. Available barrier fabrics are reported to differ extensively in both their immediate and long-term permeabilities to gases (Pullman, 1990). A study of benthic barriers (Dow Bottom Line® - a fabric that is no longer available) in the Eau Gallie Reservoir showed that barrier placement at the vegetated site was followed almost immediately by release of large quantities of gases, causing the barriers to billow up noticeably (Gunnison and Barko, 1989, 1990). In contrast, no gas collection was observed at unvegetated sites within 3 days of barrier placement and only minor volumes were collected after 8 weeks.

Gunnison and Barko, (1992) conducted laboratory studies to determine the influences of temperature, sediment type, and sediment organic matter on rates of gas evolution beneath a bottom barrier. Gas evolution was measured at 15 and 30° C from sand and clay sediments with and without additions of organic matter (plant matter). The authors concluded that problems with bottom barrier performance related to gas evolution are likely to be greatest in areas of high plant biomass. They recommended that barrier deployment be restricted to periods of the year when the standing crop of macrophytes is low. The second most important factor to consider is water temperature. Barriers should be placed during the cooler months of the year when microbial decomposition rates are low, decreasing the rate of gas release.

Bottom barriers are subject to lifting by gas bubbles from the sediments. Therefore many bottom barriers are porous or perforated to allow for gas release. However, even the most porous of materials may allow gas to accumulate. Periodic inspection of bottom barriers is required to ensure that they do not become a swimming or navigation hazard. Sometimes slits are cut into the fabric to allow gas to escape. Unfortunately, these slits can provide opportunities for aquatic plants to penetrate the barrier.

Toxicity Release of toxic materials is not expected from the use of commercial bottom barriers specifically designed for aquatic plant control or from common materials such as burlap, plastics, perforated black mylar, or woven synthetics. Routine and regular maintenance should be performed to prevent the inadvertent deterioration or loss of the barrier.

Water

Surface Water Adverse impacts to surface water quality may occur if bottom barriers are used on very large areas of aquatic vegetation. Large amounts of rapidly decaying vegetation in non-flowing water can result in oxygen depletion that can lead to fish kills. Use of bottom barriers is not expected to result in low dissolved oxygen in the water column because very large areas would need to be covered. Coverage of such areas is expected to be prohibitively expensive and it is unlikely that WDFW would issue a permit for such an extensive coverage. Ussery et al., (1997) observed a decline in dissolved oxygen to near zero beneath a bottom barrier placed in Eau Galle Reservoir, Wisconsin. This barrier also caused an increase in ammonia. Both impacts should be limited to areas covered by bottom barriers.

Another potential negative impact following bottom barrier use may be the release of organic and inorganic phosphorus during plant decomposition. Increased nutrients may result in rapid phytoplankton growth. This potential impact should not be significant if only small areas are covered.

Public Water Supplies Bottom barrier use should not disrupt public water supplies. Bottom barrier treatment creates an immediate open water column that can be sustained with annual barrier cleaning. (See Surface Water.)

Plants

Plant Habitat Bottom barriers are very effective for immediate removal of plants from the water column and can cause a 90-100 percent decrease in plant biomass. While bottom barriers cause a non-selective loss of aquatic vegetation, they are very selective for small, isolated treatment areas. Their use can have a 2-3 year or longer carryover, but plant colonization of the bottom barrier surface or from below is possible with most materials.

Helsel et al, 1996 compared 2,4-D and a bottom barrier fabric for Eurasian watermilfoil control in a Wisconsin Lake. Their objectives were to compare early-season applications of 2,4-D and bottom barriers for selective control of milfoil, regrowth of native macrophytes, and establishment of native plant beds from cuttings. They covered 675 square meters of Dunn Cove (nearly the entire area) with a polyvinyl chloride Palco® liner of 0.50 mm thickness. The bottom barrier was removed after 45 days when the underlying vegetation showed chlorosis and disintegrated easily (some coontail plants apparently survived this treatment). The site was then planted with cuttings of native submersed species. By the next summer the barrier area was dominated by Eurasian watermilfoil. The authors concluded that bottom barriers left in place for 45 days were non-selective in controlling covered plants. Replanting the area with native species proved unsuccessful, probably due to ineffective planting techniques and the drift of milfoil fragments from untreated areas. In 2,4-D treated areas, milfoil was selectively removed and native species recovered to 80 to 120 percent of their standing crop within 10 to 12 weeks after treatment.

As a matter of policy Ecology does not support removal of non-noxious emergent (wetland) species except in controlled situations where the intent is to improve low quality (Category IV) wetlands, or situations where the wetlands have been created for other specific uses such as stormwater retention.

Animals

Macroinvertebrates A study performed on a lake in Wisconsin revealed a 2/3 reduction of the benthic community after using Aquascreen® for three months (Engel 1990). Ussery et al., 1997 found that macroinvertebrate density under the bottom screens declined by 69 percent within 4 weeks of barrier placement at Eau Galle Reservoir, Wisconsin. Within a few weeks of placement at ponds near Dallas, Texas, invertebrate densities declined by more than 90 percent. Barriers also reduced macroinvertebrate taxa richness at both locations. However, biotic conditions in affected areas recovered rapidly after barrier removal. Ussery et al., 1997, noted that only macroinvertebrates directly under the barrier were negatively impacted.

Fish Sport fish forage more effectively in open areas than in plants. Bottom barriers develop their own relatively dense epibenthic fauna, which could in turn provide food. Bottom barriers would have no chronic impacts on vertebrates. However, bottom barriers can interfere with fish spawning if spawning habitat or sites are covered.

Threatened and Endangered Species Treatment with bottom barriers has the potential to affect submersed and emerged plant species federally listed as rare, threatened, or endangered. Bottom barriers are usually used only for small areas but their use does result in a non-selective loss of aquatic vegetation within the treatment area. Before the use of bottom barriers, the treatment site should be inspected for rare, threatened, or endangered species listed by US Fish and Wildlife and for "proposed sensitive" plants and animals listed by Washington State National Heritage Data System (<http://www.wa.gov/dnr/base/cons-prot.html>).

Water, Land and Shoreline Use

Aesthetics Use of bottom barriers results in decreased vegetation in small areas. This may be viewed as either a positive or adverse impact on aesthetics, depending on the attitude of the observer.

Recreation Bottom barrier use on beaches and around docks to reduce heavy vegetation is expected to improve swimming and boating activities. Steel stakes should not be used in shallow water to anchor bottom barriers because they could injure swimmers. Natural anchoring materials such as burlap sandbags or rocks are preferred. Properly maintained bottom barriers in public swimming beaches increase the safety of swimmers by allowing lifeguards to see and rescue swimmers in trouble.

Navigation Use of bottom barriers is suitable for localized control, such as around docks. To the extent that bottom barriers create small but immediate open areas of water, boat navigation would be improved after their use. Disintegration of bottom barriers into big pieces within the water column or movement of frame mounted barriers are potential dangers to navigation.

3. Mitigation, Bottom Barriers

Permits Bottom screening requires hydraulic approval that can be obtained free of charge from WDFW. If bottom barriers cost less than \$2,500, they may be exempt from the Shoreline Management Act (SMA). Barriers costing more than \$2,500 may need a Shoreline permit for installation. In any case, interested parties should check with their local government and the pertinent Shoreline Master Plan before installation of bottom barriers.

Sediment, Water, Plants and Animals Impacts from bottom barriers on sediment, water quality, plants including unique or endangered species, and animals should be minimal if used to cover a small percentage of the total bottom area of any waterbody. When there is a large standing crop of vegetation, bottom barriers should be placed in the spring before plants resume growth or in the fall when the plants have senesced. Cutting the plants prior to placement of the barrier will facilitate barrier installation, but gases will still be produced and could cause the barrier to billow.

Important fish spawning areas could be impacted if covered by bottom barriers. To avoid such impacts, the area proposed for treatment should be evaluated to determine its importance to fisheries, and critical spawning areas should be avoided. Application of bottom barriers in lakes where sockeye salmon regularly spawn requires an individual Hydraulic Project Approval (HPA) from WDFW. Application of bottom barriers in other waters may be covered by the *Aquatic Plants and Fish Pamphlet* produced by WDFW. In any event WDFW limits the area that can be covered by bottom barriers. Larger applications of bottom barriers require individual HPAs.

Impacts to federal or state listed sensitive, threatened, or endangered species (or species proposed for listing in any of these categories) could be reduced or prevented by excluding them from the treatment area. However, in order to avoid "unique" species, the location of any populations in the treatment area must be identified.

The proponent should determine if such species are in the proposed treatment area by requesting this information from Washington Natural Heritage Information System. This system provides the location of known sensitive, threatened, and endangered species populations. This data base contains only known locations so cannot be considered a comprehensive list of all locations of "unique" species in Washington. If the data system indicated that a "unique" species may exist in the project area, a survey should be conducted for field verification and the project redesigned to avoid any unique species observed (Washington State Natural Heritage Information System, 2000).

C. Suction Dredge (also called diver dredge)

Use of a suction dredge is practical for clearing plants from small areas and from areas containing obstructions, resulting in up to 90% removal. Removal can be very selective for area and for species, but increased sedimentation may obscure vision resulting in less effective harvesting.

Potential environmental impacts associated with use of a suction dredge include turbidity and re-suspension of contaminants and nutrients bound in sediment. If not identified and avoided, wetland or "unique" species may be removed. Due to the high cost of dredging and the difficulty in obtaining permits, its use and attendant impacts are expected to be confined to small areas.

1. Description

Diver dredging is a method whereby SCUBA divers use hoses attached to small dredges (often dredges used by miners for mining gold from streams) to vacuum plant material out of the sediment. The purpose of diver dredging is to remove all parts of the plant including the roots. A good operator can accurately remove target plants, like Eurasian watermilfoil, while leaving native species untouched. The operator uses a suction hose to pump plant material and sediments to the surface where they are deposited into a screened basket. The water and sediment are returned to the water column and the plant material is retained. The turbid water is generally discharged to an area curtained off from the rest of the lake by a silt curtain. Plants are disposed of on shore. Removal rates vary from approximately 0.25 acres per day to one acre per day. The suction dredge is used for small areas that require complete removal, are too large for hand removal, and are not appropriate for chemical methods. Furthermore, it can be used where bottom obstructions occur. Use of the suction dredge is slow, labor intensive, and expensive.

Diver dredging has been used in British Columbia and Washington to remove early infestations of Eurasian watermilfoil. In a large-scale operation in western Washington, two years of diver dredging reduced the population of milfoil by 80 percent (Silver Lake, Everett). Diver dredging is less effective on plants where seeds or tubers remain in the sediments to sprout the next growing season. For that reason, Eurasian watermilfoil is generally the target plant for removal during diver dredging operations.

Toxicity Release of toxic materials is not expected with use of the suction dredge. Areas offshore of storm drains should not be dredged to avoid the possibility of dredging and releasing contaminants concentrated in sediments unless these areas have been first tested using a bioassay.

2. Impacts due to Suction Dredging

Earth

Sediments Suction dredging removes roots to any depth. In flocculent sediments the plants are readily removed from the sediment. Firmer sediments may require the use of a hand tool to loosen the sediment around the roots before suctioning the plant. In hard sediments, suction dredging breaks the plant off at the roots and is not effective. Dredge use disturbs the sediments but only in very small areas of the waterbody. Discharge of the sediments back to the water column and sediments stirred up by the suction head lead to increased turbidity in the water column. The amount of turbidity present in the waterbody may be somewhat dependent on the particle size of the sediment. Fine flocculent sediments will lead to more turbidity being present in the water column following dredging.

Areas offshore of stormwater drains, combined sewer outfalls, land fills, and other areas that may contain contaminated sediment should not be disturbed by dredging to avoid the possibility of re-suspension of contaminants such as heavy metals into the water column. Dredging in such areas may release toxic materials. However, it is possible to test for contaminants using bioassay.

Air

Use of a suction dredge is expected to have little effect on air quality. Adverse effects related to its use would be associated with dredge equipment and boat or barge movement.

Water

Surface Water Suction dredging will create short-term turbidity in the water column. Dredging can also potentially release nutrients from the sediments, although impacts are expected to be short-term. Since plant materials are removed from the water immediately, decreased oxygen levels from decomposing plants are not expected to occur after treatment (See Sediments, Release of Toxic Materials).

Ground Water Suction dredge use is not expected to affect ground water.

Public Water Supplies Suction dredges may create short-term turbidity in small areas during treatment. However, public water supplies should not be disrupted by dredge use.

Plants and Animals

Plant Habitat Suction dredge use is very site specific and can be species specific. Suction dredging results in 90 percent immediate removal of plant biomass. In turbid water, a non-selective loss of vegetation may occur. Regrowth of plants in dredged areas is possible within one to two years after treatment. Suction dredging will not provide long-term control for plants that propagate by seeds, winter buds, or tubers. It is most effective for plants like Eurasian watermilfoil or Brazilian elodea which do not rely on these propagules for reproduction.

Ecology does not support removal of non-noxious emergent (wetland) species except in controlled situations where the intent is to improve low quality (Category IV) wetlands, or situations where the wetlands have been created for other specific uses such as stormwater retention.

Animals Chronic impacts on animals are not expected with suction dredge use. A slight short-term negative impact to aquatic animals may occur as a result of increased turbidity. Some substrate removal may impact benthic organisms; benthic organisms often serve as food for vertebrates. Dredging may also disturb fish spawning areas.

Threatened and Endangered Species. Treatment with a suction dredge has the potential to affect submersed and emerged plant species federally listed as rare, threatened, or endangered. Suction dredges are usually used only in small areas and can be very selective; thus impacts to threatened and endangered species are not expected.

Water, Land and Shoreline Use

Aesthetics Use of the suction dredge results in decreased vegetation in small areas. This may be viewed as either a positive or adverse impact on aesthetics, depending on the attitude of the observer.

Recreation Suction dredge use is expected to improve swimming and boating activities in areas of heavy vegetation. Fishing is not usually affected by suction dredge treatment, except that opening up areas of heavy vegetation allows anglers immediate access to fishing areas. The suction dredge is used primarily in small areas, such as for the early infestation removal of noxious aquatic weeds such as Eurasian watermilfoil and/or near obstructions such as docks. Swimming and boating should improve in areas of heavy vegetation after plant removal. Recreational facilities could be closed for short periods during dredge operation.

Navigation Suction dredge use could disrupt navigation routes during treatment. However, suction dredging is expected to improve navigation in treated areas.

3. Mitigation, Suction (or diver) Dredge

Permits Suction dredging requires hydraulic approval that can be obtained free of charge from WDFW. Generally a Temporary Modification of Water Quality Standards permit is needed from Ecology. Local agencies should be consulted to determine if any local regulations apply, often a shoreline substantial development permit is needed. In addition, the U.S. Army Corps of Engineers should be consulted to determine if a Section 404 permit is needed.

Sediment, Water, Animals, and Plants. Dredging re-suspends sediment and sediment is often discharged back to the water column after the plants are removed. Suction dredging should not be conducted in areas known or suspected to contain contaminated sediments. If contaminated sediments are suspected, sediment samples can be tested for toxicity using ceriodaphnia bioassay or other techniques before permits are issued to diver dredging projects.

Suspended sediments cause turbidity, but impacts are expected to be limited because the treatment area is generally small. If the water/sediment slurry is discharged back into the waterbody, the discharge area should be cordoned off using a silt curtain. This will minimize turbidity impacts. Diver dredging can be tailored to area and plant species unless turbidity decreases visibility. Decreased visibility makes it difficult to target specific plants, so dredging should be suspended if water becomes turbid in areas where certain plants are to be preserved. Check with the Natural Heritage Program (referenced below) to ensure that no threatened or endangered or rare plants are within the proposed treatment areas.

As with use of bottom barriers, dredging should not be conducted in critical spawning areas unless WDFW has given permission to do so. Suction dredging in lakes where sockeye salmon regularly spawn requires an individual HPA from WDFW.

D. Hand Removal, Cutting, and Raking.

1. Description

Manual methods for aquatic weed removal include hand removal, hand cutting, and raking. These methods are labor intensive and are used primarily in swimming areas and around docks. Diver hand pulling is used

increasingly to remove pioneering colonies of noxious weeds like Eurasian watermilfoil from early infestation sites or to remove plants remaining after herbicide treatments.

Toxicity Release of toxic materials is not expected with the use of manual methods of plant removal.

Hand Removal Hand removal of aquatic weeds is similar to weeding a garden. The ease and success of pulling weeds depends on the type of plant removed and type of sediment in which the plant is rooted. In water less than three feet deep no specialized equipment is required, although a spade, trowel, or long knife may be needed if the sediment is packed or heavy. In deeper water, hand pulling is best accomplished by divers with SCUBA equipment and mesh bags for the collection of plant fragments. After pulling plants from sediment, the harvester should collect all plants and fragments from the water to avoid spreading nuisance plants.

In early infestation projects, extreme care should be taken to avoid fragmentation of the plant. In some instances, a diver goody bag should be placed around the plant before pulling to catch any fragments that result. Any escaped fragments should be collected with a rake and disposed of on land. After pulling plants from sediment, the harvester should collect all plants and fragments from the water to avoid spreading nuisance plants.

Cutting Cutting differs from hand pulling in that plants are cut and the roots are not removed. Cutting is performed by standing on a dock or on shore and throwing a cutting tool into the water. Cutting generates floating plants and fragments that must be removed from water to prevent re-rooting or concentrating on nearby beaches. Weed rakes or specialized nets can be used to facilitate plant cleanup. A commercial non-mechanical aquatic weed cutter consists of two single-sided stainless steel blades forming a "V" shape. The blades are connected to a handle and to a long rope that is used to pull the cutter after it is thrown into a nuisance population of aquatic plants. As the cutter is pulled through the water, it cuts a 48-inch swath through the weeds. Cut plants rise to the surface where they can be collected and removed. Hand-held battery-powered cutters are similar to weed eaters. A long, underwater cutting blade works like a hedge trimmer to cut aquatic plants in a four-foot swath up to twelve feet below the water surface.

Raking A sturdy rake can be used to remove aquatic plants from swimming areas and around docks. Ropes can be attached to the rake to allow removal of offshore plants, and floats can be used to allow easier plant and fragment collection.

2. Impacts Due to Hand Removal, Hand Cutting, and Raking

Earth

Sediments Hand removal or raking of aquatic plants may result in some substrate removal and a short-term increase in turbidity. Increased turbidity may make it difficult to see remaining plants and may disturb benthic organisms. The degree of turbidity will depend on the type and texture of the sediment, the density of the plants being removed, and the depth of the plant roots. Removal of dense plant beds may change the flow rate and sedimentation rate in flowing waters (this holds true for all the other methods too).

Water

Surface Water Hand removal and raking of aquatic vegetation may result in increased turbidity in limited areas during treatment. If pulled or cut plants are removed from the water, increased nutrients and/or decreased oxygen levels are not expected to occur in the treated lake or pond; however there may be some increase in nutrients due to sediment re-suspension. These effects are expected to be short-lived.

Public Water Supplies Manual methods (especially hand-pulling of plants) may result in a short-term turbidity increase in the treatment area.

Plants and Animals

Plant Habitat Hand pulling can be species specific in removal of aquatic vegetation with a minimum disruption of native plants. It is more difficult to target specific species during raking or cutting activities. It is hard to collect all plant fragments using manual methods, some species are very difficult to uproot with manual methods, and treatment may be required several times each summer. Because it is so labor intensive, manual plant removal is not practical for large areas or for thick weed beds.

Ecology does not support removal of non-noxious emergent (wetland) species except in certain situations where the land managers plan to improve low quality wetlands (Category IV) and in wetlands created for other specific uses such as stormwater retention.

Animals Hand removal of aquatic plants disturbs benthic organisms. Since manual methods are slow and labor intensive, removal of an entire lake plant community is not expected. Therefore habitat for other aquatic organisms (such as fish) is not expected to be greatly impacted by the use of manual methods.

Threatened and Endangered Species Manual methods of aquatic plant removal have the potential to affect submersed and emersed plant species federally listed as rare, threatened, or endangered. However manual methods can be species specific in removal of plants and are generally used for small areas so if identified, these species can be avoided.

Before manual methods are used for plant removal, each site should be reviewed for rare, threatened or endangered species listed by US Fish and Wildlife and for "proposed sensitive" plants and animals listed by Washington State National Heritage Data System.

Water, Land and Shoreline Use

Aesthetics Manually removing vegetation from small areas may be viewed as either a positive or adverse impact on aesthetics, depending on the attitude of the observer.

Recreation Manual removal of plants on beaches and around docks is expected to improve swimming and boating activities. Fisheries are not expected to be affected by manual treatment of relatively small areas of aquatic vegetation.

Navigation Use of manual methods is suitable for localized control, such as in swimming areas and around docks. Small open areas of water which result from manual method use will improve boat navigation.

3. Mitigation, Manual Methods

Permits Handpulling, raking, and cutting (including battery-powered equipment) requires an HPA from WDFW. Manual methods in lakes where sockeye salmon regularly spawn requires an individual HPA from WDFW. Manual techniques in other waters may be covered by the *Aquatic Plants and Fish Pamphlet* produced by WDFW. In any event, WDFW limits the area of aquatic plants that can be removed by manual methods.

Sediment, Water, Animals, and Plants Small scale manual methods would minimally impact these elements of the environment. Nevertheless, care should be taken to avoid unique plant species and critical fish spawning areas.

E. Rotovation

Rotovation is performed using agricultural tilling machines that have been modified for aquatic use, or machines that have been specially designed for rotovation. Rotovators use underwater rototiller-like blades to uproot aquatic plants. Rotating blades churn seven to nine inches deep into the lake or river bottom to dislodge plant roots. Plant roots are generally buoyant and float to the surface of the water. Generally, rotovators are able to extend 20 feet under water to till substrate, and may be able to till shallow shoreline areas if access is not limited by the draft of the machine. Rotovators do not collect roots and plant fragments as plants are uprooted. However, plants and roots may be removed from the water using a weed rake attachment to the rototiller head, by harvester, or manual collection. In Washington and British Columbia, rotovation is primarily used to remove Eurasian watermilfoil from lakes and rivers. Rotovation was also used to successfully remove water lily (*Nymphaea odorata*) rhizomes from a lake near Seattle. Rotovation appears to stimulate the growth of native aquatic plants, so it would probably not be an effective tool to manage excessive growth of nuisance native species.

The optimum time for rotovation extends from late fall to spring. During this period, plant biomass is reduced as is the number, buoyancy, and viability of plant fragments; water levels; and conflicts with beneficial uses of the water body (Gibbons, Gibbons, Pine; 1987). Due to increased plant biomass during summer months, plants must be cut before rotovation. Otherwise the long plants tend to wrap around the rototilling head.

The area that can be rotovated per day can range from 2 acres to less than 1 acre depending on plant density, time of year, bottom obstructions, plant species, and weather conditions. Generally, rotovators are not able to operate efficiently in winds over 20 miles per hour. Imprecise tracking of treated areas may result in incomplete removal of target plants, ultimately reducing long term-control. Tracking efficiency can be improved with use of buoys.

Rotovation can effectively control milfoil for up to two seasons. Deep-water rotovation has resulted in an 80% to 97% reduction of milfoil, with control lasting up to two years. The rotovated area is eventually recolonized by milfoil fragments that float in from untreated areas or from plants remaining after rotovation.

Potential significant environmental impacts associated with rotovation include increased sedimentation, re-suspension into the water column of sediment-bound contaminants, and surface water contamination from spills of hydraulic fluid or fuel. Rotovation is not selective within the treatment area and could result in removal of desirable species such as wetland vegetation or "unique" species. However, removal of

monotypic vegetation such as milfoil may ultimately increase diversity of desirable species and rotovation appears to stimulate the growth of native aquatic plants. Rotovation temporarily disrupts the benthic community, which in turn could impact benthic feeders.

Use of rotovators can result in plant fragments. If not collected, decaying plant fragments could reduce dissolved oxygen levels and increase nutrients. Plant fragments could also clog water intakes and trash racks of dams, and may result in increased dispersal and colonization of some species (including Eurasian watermilfoil). Rotovation should be used only in waterbodies where Eurasian watermilfoil fully occupies its ecological niche. Otherwise rotovation could tend to spread Eurasian watermilfoil throughout the waterbody rapidly. As discussed in the "Impacts" section, mitigation measures could be designed to reduce or avoid some of the impacts discussed.

Several permits and compliance with the State Environmental Policy Act are required prior to rotovation. Local jurisdictions (cities, counties) may require a shoreline permits, Ecology requires a temporary modification of water quality standards issued by the regional offices, and a Hydraulic Project Approval is required from WDFW. In addition the U.S. Army Corps of Engineers requires a section 404 permit.

2. Impacts Due to Rotovation

Earth

Sediments The rotovator's tiller head can penetrate sediment to a depth ranging from 7 to 9 inches. Rotovation re-suspends sediments, resulting in turbidity and increasing the potential for re-suspending toxic substances. Depending on sediment consistency (muck, sand, etc.) and density of the root mass, root removal may increase the amount of sediment re-suspended and the depth to which sediment is disturbed (Moore, A. Personal communication.). Sediments in the treatment area could be contaminated with metals, pesticides, or other toxic substances as a result of historical or existing uses. Sediments may also contain high levels of nutrients, which if re-suspended could fuel phytoplankton blooms.

Sediment disruption may cause movement of contaminants, either to the sediment surface or into the water column. Standards have not yet been set for fresh water sediments so it is difficult to assess benthic impacts, which would vary depending on the type and concentration of contaminant. The Lake Osoyoos Rotovation Demonstration Project (Gibbons, Gibbons, Pine, 1987) characterized surficial sediment quality before and 2.5 months after rotovation. Lake Osoyoos was chosen as the study site for rotovation because land use practices made it likely to have sediments contaminated with heavy metals and pesticides. In most cases, where metals were detected before treatments, levels were somewhat elevated after treatment. Bis (2-ethylhexyl phthalate) concentrations were dramatically higher after treatment (<330 ppm before, 4,400 ppm after).

Gibbons et. al. 1987, concluded that there was no apparent effect from rotovation on the limited number of species comprising the benthic community in Lake Osoyoos. However, data indicate that species shifts did occur and that there was a post-rotovation reduction in diversity of benthic species. This reduction was most noticeable two months after rototilling but still in evidence 5 months later.

Water

Surface Water (see also, sediment section) Lake Osoyoos Rotovation Demonstration Project researchers concluded that rotovation may have minimal impacts on water quality (Gibbons, Gibbons, and Pine; 1987).

However, study results may not have been conclusive because the rotovator periodically malfunctioned, resulting in less intensive tilling and thus less disruption of sediment. Researchers found that rotovation did not alter dissolved oxygen levels, pH, or water temperature. Rotovation caused temporary turbidity, and phosphorous levels were slightly elevated for the first 24 hours after treatment.

Water quality samples taken before, during, and after rotovation were sampled for pesticides and 13 metals. Copper, nickel, and zinc were the only metals above detection levels in any sampling period. Concentrations of copper and nickel showed a minimal increase after treatment, however the level of zinc in the drift zone exceeded Chronic EPA Freshwater Biota Criteria. The high level of zinc in the drift zone may be linked to rotovation, indicating a potential for adverse impacts to water quality from rotovation. Additional research would be required to accurately characterize the potential impacts of sediment disturbance from rotovation on water quality. Since impacts could vary dramatically among rotovation sites, impacts should be assessed for each proposed treatment site. Lake Osoyoos was chosen as the study site because it represents a worst case scenario for heavy metals and pesticides due to land use practices around the lake.

Incidental loss of hydraulic fluid or other petroleum products may also impact water quality. If fluid lines are not maintained and proper care not taken when changing equipment such as cutter heads, the number of incidents of release of petroleum products to surface water could be high although the amount of fluid lost each time may be moderate (~5 gallons). If equipment were not maintained the amount of fluid lost could be much greater (~50 gallons), particularly if hoses were not equipped with shut-off valves (Cornett and Hamel, Personal communication. 1991). Also, in-water disposal of plant fragments could result in reduced dissolved oxygen levels as plant matter decomposes, potentially resulting in fish kills.

Cut plants leak nutrients back to the water column, generally within one hour of being cut. Unless a plant harvester immediately harvests cut plants, some plant nutrients would enter the water.

Water Supplies If cut plants were not removed from the water after treatment, fragments could clog water intakes. In addition, rotovation itself may damage individual water intake pipes. Water supplies could be impacted by turbidity or re-suspended contaminants. The potential for and level of impacts would depend on the proximity of an intake to disturbed sediments and the amount and toxicity of re-suspended contaminants. See "sediment" section.

Plants and Animals

Plants Rotovation has resulted in a 80% to 97% reduction of Eurasian watermilfoil stem density with control lasting up to two years (Gibbons, Gibbons, Pine, 1987; Hamel, Personal communication.). Rotovation has been shown to alter species composition and increase species diversity of desirable plant species. Removing milfoil and rototilling appears to stimulate seed germination and growth of native species (Hamel, K. Personal communication.). Rotovation is not selective within the target area, therefore any desirable species in the target area, including wetland species, would be removed on a temporary basis.

Animals Removal of desirable plant species may eliminate valuable habitat for a variety of animal species. However rotovation of milfoil increases plant species diversity, which enhances habitat.

Information available on the impacts of rotovation on fish is inconclusive due to the lack of an accurate method for assessing impacts on fish populations in Eurasian watermilfoil beds (Gibbons, Gibbons, Pine, 1987; Coots, R. Personal communication). Some disturbance of behavioral patterns could be expected, particularly if spawning or rearing areas were disturbed. Impacts would depend on species using the water

body, habitat value of plants removed, and level of disruption. In the long term, rotovation to remove Eurasian watermilfoil may benefit fish by removing a monotypic species and replacing it with a diverse native community. In British Columbia, rotovation has been used to remove Eurasian watermilfoil from salmon spawning beds that had been invaded, thus returning them to use by salmon.

Threatened and Endangered Species Rotovation is not selective. Any sensitive, threatened, or endangered plant species within the treatment area would be temporarily eliminated. However, both the rotovation process and removal of milfoil from an area appear to have a stimulatory effect on native aquatic plants. Native plants may prosper after rotovation.

Energy, Transportation, and Natural Resources

Rotovation above dams could interfere with power generation if plant fragments were allowed to clog trash racks of dams (Hamel, K. personal communication). Eurasian watermilfoil does produce fragments on its own and these naturally produced fragments also impact dams.

3. Mitigation, Rotovation

Permits WDFW requires an HPA prior to rotovating and before deadheads or logs can be removed and in many cases will not allow woody debris to be removed from a waterbody. Ecology requires a permit, counties and cities sometimes require a shoreline permit, and the Army Corps of engineers may require a Section 404 permit.

Water/Sediment Quality A review of historical and current use of the proposed treatment area may be required to help determine if contaminants exist in sediments in the treatment area. Should this or other information indicate that sediments may be contaminated, permittees may require a sediment bioassay on suspected sediments prior to issuing a permit for rotovation. Work in or near the waterway should be done so as to minimize streambed erosion, turbidity, or other water quality impacts. Maintenance and operation procedures performed on rotovation equipment could release petroleum products or other toxic or deleterious materials into surface waters. Thus, such procedures may be required to be conducted at upland locations to prevent entry of toxic substances into waters of the state.

Due to the high probability of hydraulic fluid or fuel leakage into state waters caused by equipment failure or poor maintenance, permittees may require a detailed inspection plan complete with maintenance logs to be kept and available for inspection. Additionally, operators may be required to complete a daily inspection of all hydraulic equipment, fuel systems, and other systems that may cause petroleum products to be discharged to waters of the state. Permittees may also require that no extra fuel or hydraulic oil be kept on board the rotovator in excess of the amount necessary for emergency repair or re-fueling. To minimize impacts should a spill occur, operators may be required to carry on board the rotovator at all times oil-spill materials such as a containment boom and absorption pads. They may be required to develop a spill contingency plan. The hydraulic system of rotovators should be upgraded to operate only on food grade oil only.

To avoid impacts associated with plant fragments, the applicant may be required to dispose of vegetation on land in such a manner that it cannot enter into the waterway or cause water quality degradation to state waters.

Public Water Supplies To avoid damage to water intake pipes, individuals should be given adequate notice of the treatment, informed of the potential for damage to intake pipes, and asked to pull intakes from the water prior to treatment.

Plants and Animals Ecology does not support removal of non-noxious emergent (wetland) species except in controlled situations where the intent is to improve low quality (Category IV) wetlands or situations where wetlands have been created for other specific uses such as stormwater retention. Areas containing desirable species, such as emergent wetland species, should be avoided.

An evaluation of each proposed treatment site should be required to determine if the site is used by fish for spawning, rearing, or other purposes. If the area does provide important habitat, the proposal should be designed to avoid impacts, either by avoiding or limiting the treatment area, or scheduling treatment to avoid interference with critical uses. Turbidity and disturbance caused by rotovation may interfere with juvenile salmon or fish passage. Therefore, WDFW imposes timing restrictions on when rotovation may be allowed to occur within each waterbody. Because timing restrictions have been severe in salmon-bearing waters and because rotovation is extremely expensive, it has not become a popular method of aquatic plant control in Washington.

F. Mechanical Cutting and Harvesting

Mechanical cutting and harvesting are practical for large-scale (several acres) vegetation removal because they remove plants from large areas in a relatively short time. Regrowth may occur within one month after cutting or harvesting; therefore several treatments per season may be required. While these methods may be useful for control of aquatic vegetation, they would not result in total eradication of noxious species such as Eurasian watermilfoil.

Use of these methods has the potential to result in some significant adverse environmental impacts, but impacts would generally occur within the target area. Mechanical cutting and harvesting may disturb sediments but only if the equipment is operated in areas too shallow for the cutter setting. Mechanical cutting and harvesting are non-selective and could eliminate valuable fish and wildlife habitat within the target area. Generally some plant biomass remains in the water and is available as habitat. Additionally, research indicates that operation of mechanical harvesters can kill up to 25% of small fish in a given treatment area.

Use of cutters, and harvesters to a much lesser degree, can result in accumulation of plant fragments. If not collected immediately, decaying plant fragments can reduce dissolved oxygen levels and increase nutrients. Cut plants leak nutrients back into the water column within one hour of being cut. Plant fragments could also clog water intakes and trash racks of dams, and may result in increased dispersal and colonization of some species. Disposal of fragments is another consideration.

1. Description

Mechanical harvesters are large specialized floating machines that cut, collect, and store plant material. Cut plants are removed from the water by a conveyer belt system and stored on the harvester until removed for disposal. A barge stationed near the harvesting site for temporary storage is an efficient storage method; alternately the harvester carries cut plants to shore. Cut plants may be disposed of in landfills, used as compost, or used to reclaim spent gravel pits or similar sites.

Harvesting is usually performed in late spring, summer, and early fall when aquatic plants have reached or are close to the water's surface. Harvesters may operate every day throughout the growing season, particularly if the treatment area is large. Harvesters can harvest several acres per day depending on plant type, density, and harvester storage capacity. Depending on the equipment used, plants are cut from 5 to 10 feet below the water surface in a swath 6 to 20 feet wide. Because of the large machine size and cost, harvesting is most efficient in water bodies larger than a few acres. Harvesting can be used as a nutrient

removal technique because the cut plants are immediately removed from the water and disposed of off-site. Thurston County performs a fall harvesting to remove senescing plants and their nutrients from the Long Lake. Harvesting can be a nutrient management technique in swallow eutrophic systems.

Mechanical Plant Cutters Two commercial types of mechanical underwater plant cutters are available. Portable Boat Mounted Cutting Units are portable boat-mounted cutters that can be installed on a 14 foot or longer boat and is capable of cutting a 7 foot swath four feet below the waters surface at a rate of about one acre per hour. Specifications may vary depending on the manufacturer of the equipment.

Specialized Barge-like Cutting Machines are mechanical cutters similar to harvesters but differ in that cut plants are not collected as the machinery operates. These machines can cut plants in water as shallow as 10 inches and as deep as 5 feet, with the main sickle cutting a 10 foot wide swath. Specifications may vary depending on the manufacturer of the equipment. Specialized barge-mounted cutters can cut up to 12 acres of plants per day in open water. Cutting is generally performed during the summer when plants have reached or are close to the water surface.

Effectiveness of mechanical harvesting and cutting for controlling aquatic vegetation depends on depth of cut from surface and bottom, time of year, plant density and biomass, distance to off loading sites, cutting speed of the equipment, and the number of cuts per season. Literature specific to Eurasian watermilfoil identifies the proximity of the cutter head to milfoil root crowns as a factor-influencing efficacy. Harvesting and cutting can interfere with carbohydrate allocations from roots and shoots, which in turn can weaken the plant making it more susceptible to natural controls (Gibbons, 1986). It can also affect storage of nutrients so that it may not over winter as well and may not grow as vigorously the following year (Hamel, K. 1991).

Cutting and harvesting both result in immediate areas of open water; however, two or three treatments per season may be required to maintain open water. Cutters are smaller than harvesters and are generally more maneuverable allowing for plant removal around docks, boat moorages, and restricted areas.

2. Impacts Due to Mechanical Harvesting and Cutting

Earth

Sediments Incidental sediment disturbance may occur if blades on barge-mounted mechanical cutters are set too deep. Paddle wheels on some mechanical harvesters may re-suspend sediments (Engel, 1990). If cutters or harvesters disturb contaminated sediments, contaminants could be released into the water column, with the potential impact depending on the toxicity and amount of contaminant released.

Collected plants must be disposed on land, which requires off loading sites to be identified. Adverse impacts to the shoreline may occur as heavy equipment is used to remove cut plants from the harvester. The plants must be disposed in landfills or can be used for compost.

Water

Temporary turbidity could result if sediments were disturbed. If cut plants were not removed from the water, decaying plant material could deplete dissolved oxygen levels and increase nutrients. Also, uncollected plant fragments could clog water intake systems.

Plants and Animals

Plants Mechanical cutters and harvesters are not selective within the target area; therefore any desirable species within the target area may be cut and collected. Uncollected plant fragments may increase dispersal and colonization of noxious species such as Eurasian watermilfoil. Some plant fragments escape even the best of harvesters. These plant fragments may drift into other parts of the waterbody and take root, while others may wash up on shore.

Mechanical harvesting could affect the composition of plant communities (Engel, 1990). After harvesting in a Wisconsin Lake, vegetation was altered from a predominant mix of coontail, Berchtold's pondweed, curly-leaf pondweed, and sago pondweed to a 6-year dominance by water star grass. Generally plants that reproduce sexually, regenerate poorly from cut parts, heal and regrow poorly when cut, and are tall are most vulnerable to harvesting (Nicholson, 1981). These characteristics fit many native species, especially the pondweeds (*Potamogeton* spp.). Plants like Eurasian watermilfoil may be favored by harvesting. In Lake Wingra Wisconsin, Stanley et al, 1994, compared areas with a history of mechanical harvesting to other areas with no known management history. Although species diversity and taxa richness in three out of four unharvested areas were greater than in the harvested area, no differences in diversity of plant biomass could be attributed solely to the harvesting regime.

Harvesting has been used extensively in Lake Minnetonka, Minnesota, to control Eurasian watermilfoil. Crowell et al., 1994 measured effects of harvesting in five locations in Lake Minnetonka and reported that the relative growth rates of plants in the harvested area were greater than in adjacent unharvested plots. However, the increased growth rate did not result in greater canopy density or higher total shoot biomass in the harvested areas. Harvesting also reduced the plant abundance at the water surface for up to 6 weeks following the harvest, when harvested in early July. Other researchers have found that harvesting reduced biomass for only 3 to 4 weeks (Cooke et al., 1990). Seasonal timing of harvesting may affect the duration of control

Animals Reduction of desirable plants from the upper water column through harvesting or cutting may remove habitat used by animals and waterfowl for wintering, breeding, rearing, nesting, and feeding, as well as alter migration routes. The severity of impact would depend on the value of habitat removed and location (i.e. proximity to flyways, migration routes, etc.). Physical intrusion may alter animal behavior, although information related to this impact was not available.

Mikol 1985 estimated that 2226-7420 fish per hectare were removed by conventional harvesting of plant beds dominated by Eurasian watermilfoil. Similar removal rates were observed in a two-year Wisconsin study where mechanical harvesting of 50 to 70% of submersed plants in Halverson Lake killed 2100 fish per acre harvested, or about 25% of all fry in the lake (Engel, 1990). Because adult fish are more able to flee or avoid the treatment area, impacts on adult fish were less than those on fry. Other factors found to influence the number of fish killed were the number, size, and location of fish, and harvester handling. In some lake systems, especially those with an overabundance of aquatic plants, removal of juvenile warmwater fish such as bluegills may actually improve the fishery.

This Wisconsin study also found that harvesting resulted in a loss of 22% (in June) and 11% (in July) of all plant-dwelling macro invertebrates in the lake. Patches of displaced snails, caddisfly larvae, and chironomids drifted about Halverson Lake and onto shores after harvesting. Both bass and bluegills were seen devouring insects dislodged during harvesting. Harvesting had a minimal effect on phytoplankton.

In a 1996 harvesting study on Lake Keesus, Wisconsin, Booms estimated that annual harvesting operations removed about 39,000 fish from this lake. Bluegills between 4 and 10 cm in length were the most common fish removed comprising 46 percent of the fish taken. Others included largemouth bass (24 percent), unidentified fry (16 percent), and black crappie (8 percent). Generally smaller fish were removed. Mud puppies, adult and immature bullfrogs, and larger fish (12 – 56 cm long) were occasionally harvested during normal harvesting operations. Booms estimated that approximately 700 turtles were also removed during the 1996 harvesting season.

The native weevil (*Euhrychiopsis lecontei* Dietz) has been proposed as a possible biological control for Eurasian watermilfoil. Sheldon and O'Bryan, 1996, investigated impacts of a harvesting program on weevil densities in Lake Bomoseen Vermont. They found that there was a significant negative effect of weed harvesting on weevil abundance. There were fewer weevils found in the harvested sites, whereas weevil densities in unharvested sites remained higher. Milfoil weevils spend most of their time in the 1.5 m apical portion of plants which is the part of the plant removed by the harvester.

Threatened and Endangered Species Mechanical cutting and harvesting is not selective. Any sensitive, threatened, or endangered plant species within the treatment area would be cut and collected. Cutting a plant does not necessarily eliminate it. Care should be taken to avoid harvesting threatened or endangered plants.

A harvesting operation could remove juvenile salmon from plant beds. Harvesting operations in salmon bearing waters should be carefully evaluated before permits are issued to harvest.

Water, Land and Shoreline Use

Recreation Swimming, fishing and other forms of recreation should be restricted in areas in which cutters or harvesters were operating to avoid danger to recreationalists. Generally harvesting and cutting operations open up large areas of water and provide better recreational opportunities for swimming, boating and fishing. Using harvesters to cut fishing lanes can increase fish and fishing productivity by providing plant bed edges. Fish, such as bass, can target smaller food fish and anglers have better fishing access in such areas.

3. Mitigation, Mechanical Harvesters and Cutters

Permits Harvesting in Washington requires an HPA from WDFW. Some Shoreline Master Programs may also require permits for harvesting. Check with your city or county government.

Sediment To minimize sediment disruption, operators may be required to insure that the depth of mechanical cutter blades and harvester wheels would not extend into the sediment. Operators may be instructed to limit activities to waters more than five feet deep or so.

Water Operators may be required to remove all cut plants from the water so as to avoid impacts to water quality and public water supplies.

Plants and Animals To avoid impacts related to loss of habitat, a survey of each area proposed for treatment may be required to determine habitat value of plant species, and the potential impact of plant removal. Survey results would dictate appropriate mitigation, which could include limiting the size or location of the harvest area, and/or extent of the harvest. Proponents may be required to design the project to avoid migration routes, critical habitats, including wintering, breeding, rearing, nesting, and feeding habitats. The duration of control may be lengthened by harvesting later in the season (July instead of May or June).

To minimize fish losses, operators may be required to remove fish as plants move up the harvester conveyor belt. Fish loss may also be reduced or prevented by altering the harvest schedule to accommodate fish spawning, rearing, or other behavior. For example, if fry use near-shore areas in early summer, harvesting of these areas could be delayed until fry moved out of the treatment area. Thurston County specifically avoids harvesting areas of thin-leaved pondweeds because they found that these areas support large populations of fish. Appropriate mitigation may require assessment of species use and behavior in the proposed treatment area.

Areas should be set aside for conservation where the milfoil eating weevil *Euhrychiopsis lecontei* is present and desired as a biological control for Eurasian watermilfoil. These areas could include shoreline areas where there was no human activity or in areas where harvesters could not effectively cut (extensive shallow areas).

Impacts to federal or state listed sensitive, threatened, or endangered species (or species proposed for listing in any of these categories) could be reduced or prevented by excluding them from the harvest area. However, in order to avoid "unique" species, the location of any populations in the treatment area must be identified.

At a minimum, the applicant could be required to provide verification of a search of the Washington Natural Heritage Information System, which provides the location of known sensitive, threatened, and endangered species populations. This data base contains only known locations, so cannot be considered a comprehensive list of all locations of "unique" species in Washington. If the data system indicated that a "unique" species may exist in the project area, a survey should be conducted for field verification, and the project redesigned to avoid any unique species observed.

The proponent may be required to establish setbacks from breeding sites, nests, and feeding or perching areas for federal and state sensitive, rare, threatened, endangered, or unique species and species proposed for listing as such.

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Section VI. Alternative 5 – Biological Methods Only

Introduction to Biological Controls

Under this alternative, agencies process permits or funding allowing the introduction of sterile grass carp (*Ctenopharygodon idella*) into waters of the state. Other biological methods reviewed in this EIS, including plant pathogens, herbivorous insects, competitive plants, and plant growth regulators, are not yet realistic alternatives. Many of these options appear to be promising alternatives for aquatic plant control and may be considered after undergoing further laboratory and field analysis.

A. Plant Pathogens

Preliminary research has demonstrated that plant pathogens may be useful in the future control of aquatic vegetation in general and hydrilla and Eurasian watermilfoil in particular. The establishment of inoculation strategies and inoculum thresholds and determination of the optimum time in the hydrilla and Eurasian watermilfoil life cycle for initiation of infection are some topics requiring further research. The use of plant pathogens in conjunction with mechanical techniques or with organisms that physically damage plant tissues to provide inoculation sites may be particularly effective (Gunnar 1983). Recent research shows that using fungal pathogens in conjunction with low levels of aquatic herbicides is particularly effective in managing problem plants in the laboratory.

In the mid-eighties, a survey of the continental US for pathogens of Eurasian watermilfoil was conducted on more than 50 waterbodies in 10 states (Zattau 1988). Bacteria isolates (462) and fungal isolates (330) were collected and maintained in pure culture. Lytic enzyme assays indicated that 36 isolates had potential as biocontrol agents; further assays indicated 5 fungal isolates which may be particularly effective after additional study.

At this time, the most promising plant pathogen as a biological control agent for Eurasian watermilfoil and hydrilla is the fungus *Mycoleptodiscus terrestris* (Winfield 1988). Extensive research on this fungus is underway in a number of laboratories and is described below. A rapid and devastating response by watermilfoil to the fungus plus associated bacteria was observed in laboratory experiments; field experiments using only the associated microorganisms demonstrated that they may provide ecocites for the fungus by pitting the plant surface (Gunnar et al. 1988).

Further research on plant microbe interactions, the phase at which specific association may occur, and host specificity to two fungi was recently reported (Kees and Theriot 1990). Using a different approach, Stack (1990) constructed an epidemiological model that described the interaction of an aquatic plant host with a fungal plant pathogen using *M. terrestris* as the fungal agent and watermilfoil as the host. Currently, Winfield (1990) is investigating the optimum shelf life and optimum level of *M. terrestris* inoculum needed for biocontrol of watermilfoil. Finally, Andrews et al. (1990) recently assayed microbial colonization of Eurasian watermilfoil by other fungi.

B. Herbivorous Insects

Further laboratory and field research needs to be conducted before herbivorous insects are available for use in aquatic vegetation control. Researchers from the US Department of Agriculture are currently surveying waters in China for potential biological control agents for Hydrilla and Eurasian watermilfoil (Balciunas 1990).

In British Columbia, researchers have observed several species of aquatic insects grazing on Eurasian watermilfoil (Kangasniemi and Oliver 1983). The chironomid larvae *Cricotopus myriophylli* showed particular promise as a biological control agent. This insect effectively reduces the height of watermilfoil plants by feeding on meristematic regions. *C. myriophyllum* prefers *Myriophyllum spicatum* over *M. exalbescens* (a native watermilfoil species). It is likely that *C. myriophylli* has spread downstream into the US through the Columbia River systems. Further research is needed to determine how to produce or sustain insect populations to attain effective control and to determine when the target plant is most vulnerable to attack. Development of techniques for adult mating and egg collection remains the most critical limitation to laboratory rearing.

In Vermont in the 1980's, Eurasian watermilfoil populations in Brownington Pond were significantly decreased by several underwater insects. Researchers believe declines could be due to either two aquatic caterpillars (*Acentria nivea* = *A. niveus* and *Parapoynx* sp.) or an aquatic weevil (*Eurhynchiopsis lecontei*) (Sheldon 1990). The goal of future work is to evaluate the potential of one or more of the herbivorous insects to control watermilfoil in other lakes.

Creed et al. added weevils (*Eurhynchiopsis lecontei* Dietz) to *Myriophyllum spicatum* growing in laboratory aquaria. After harvest it was determined that some of the aquaria also contained the aquatic caterpillar (*Acentria nivea*), so effects were attributed to herbivory in general. Both the weevil and caterpillar expose stem vascular tissue when feeding and this leads to the collapse of milfoil plants from the water's surface. The authors concluded that these herbivores do not have to remove considerable amounts of stem or leaf tissue in order to have a strong negative effect on milfoil. A collapsed plant sinks from the well-lit surface waters, sometimes carrying undamaged plants with it. Milfoil plants may not be able to get enough light for photosynthesis at these lower depths." From management viewpoint a collapsed plant is also off the surface and causing less impact to recreation and aesthetics.

A number of weevil augmentation experiments have been conducted where numbers of laboratory-reared weevils were introduced into lakes in Vermont and the Mid-west. Results have been mixed, with declines in Eurasian watermilfoil in some waterbodies and no declines in others. Factors governing weevil densities are still unclear, but this method shows great promise as a biological control for Eurasian watermilfoil.

Ecology is funding research at the University of Washington to evaluate whether the milfoil weevil will be a suitable control for Eurasian watermilfoil in Washington. Unfortunately, densities of these naturally occurring native weevils in Washington appear to be much lower than the natural densities seen in other states. In comparison to states where weevils have been observed causing declines, Washington has cooler summer water temperatures.

C. Competitive Plants

Interspecific competition may be an effective aquatic plant control method in some situations. Further research is needed to determine specific conditions which enable native plant species to outcompete invasive species such as purple loosestrife or Eurasian watermilfoil.

In a 1986 study, researchers investigated the establishment of spikerush (*Eleocharis coloradoensis*) following chemical control (2,4-D) of watermilfoil and showed mixed results (Gibbons et al. 1987). Spikerush was successful in surviving and reproducing in shallow areas planted with large, densely populated strips of cut sod. However, it was not successful in areas planted with strips composed of small wet plugs. Wave and water circulation patterns played a major role in transplant success.

D. Plant Growth Regulators

A new strategy for aquatic plant management involves the use of plant growth regulators. These compounds inhibit gibberellin synthesis, thereby inhibiting normal plant elongation. Early research in the

laboratory resulted in a bioassay system using hydrilla and Eurasian watermilfoil (Lembi et al. 1990). The bioassay suggests that gibberellin synthesis inhibitors uniconazol, flurprimidol, and paclobutrazol were effective in reducing plant height in aquatic systems but would have minimal adverse impacts on plant health (Lembi and Netherland 1990). (Note: Although plant growth regulators are chemical control methods, they are included in the biological section because they are natural chemicals, not synthetic. They will require further research as will plant pathogens and herbivorous insects before they are ready for commercial use.)

E. Mitigation: Plant Pathogens, Herbivorous Insects, Competitive Plants, Plant Growth Regulators

As noted in the section describing biological methods and their impacts, additional research and licensing must be conducted before using plant pathogens, herbivorous insects, competitive plants, and plant growth regulators. Appropriate additional environmental review will be conducted once these methods become available.

F. Grass Carp

Washington Department of Wildlife (WDFW) evaluates use of grass carp use in Washington (See Appendix E). Ecology has included grass carp as part of the integrated management approach of the Aquatic Plant Management Program, but all requests for game grass carp stocking and planting permits should be made to WDFW.

1. Description

The grass carp, also known as the white amur, is a fish native to the Amur River in Asia. Because this fish feeds on aquatic plants, it can be used as a biological tool to control nuisance aquatic plant growth. In some situations, sterile grass carp may be permitted for introduction into Washington waters.

Grass carp are a member of the minnow family. Grass carp can grow to 100 pounds in their native home range and can live for more than 20 years. Grass carp's natural habitat includes the large, swift cool rivers of China and Siberia. However, all grass carp in the United States are of Chinese origin (Pauley and Bonar). Female grass carp usually reach sexual maturity a year ahead of males, and the age of maturity depends on climate and nutrition. Female size at maturity is usually five to ten pounds, and the average ten to 15 pound female will produce 500,000 eggs each year. Water temperatures ranging from 59 - 63° F trigger upstream migration to spawning grounds where grass carp spawn from April to August or September. Depending on temperature, eggs hatch in 16 to 60 hours, are free floating, and drift with the current. Newly hatched larvae absorb their yolk sacs at about one-third inch long and begin feeding on plankton; however, at one inch the fry start feeding on aquatic vegetation. Small grass carp prefer tender, succulent plants, and as the fish grow their preference range for aquatic plants broadens.

Grass carp have special teeth in their throats and a horny pad that enables them to cut, rasp, and grind aquatic plants which ruptures the plant cell membranes to allow digestion of plant material. Grass carp do not pull plants up by the roots like the common carp but eat from the top down without disturbing roots or sediment.

Intensive feeding begins at water temperatures above 68° F, while feeding diminishes below 53° F. Dissolved oxygen levels less than four ppm also reduce food intake by as much as 40 percent. Grass carp can consume up to 150 percent of their body weight per day when temperatures are above 77° F but below 90° F. Grass carp can survive a wide range of temperatures from freezing to 95° F. They cannot survive in salt water but can migrate through brackish water. Growth rates of triploid grass carp were studied from four

Washington lakes. Growth was highest in East Pipeline Lake where grass carp grew from an average of 144 grams to 6032 grams in approximately 4.3 years. In approximately the same time period, two size classes of grass carp grew from an average of 144 grams and 732 grams to 4419 grams in Keevies Lake and from an average of 144 grams to 3701 grams in Bull South Lake. In Big Chambers Lake, two size classes of grass carp grew from 223 grams and 282 grams to 2363 grams in approximately 1.3 years. Triploid grass carp growth rates in this study compared favorably to growth rates of grass carp from similar climatic areas and were equal or greater than growth rates of grass carp from their native range (Pauley and Bonar).

Grass carp were first brought to the U.S. in 1963 in Arkansas and other southern states. Fertile, diploid grass carp were stocked in initial treatments and because of the unknown potential impact to native fish and wildlife species, many states prohibited their use. They were declared deleterious exotic wildlife by WDFW in 1973. By the early 1980's, triploid grass carp, which are sterile, were being produced in the U.S. Researchers in regions where grass carp rapidly reach maturity have concluded that triploid fish are "functionally sterile". The hatching success of triploid x triploid crosses is less than 0.5 percent and all of these offspring are triploid. Normal diploid hatching success ranges from 40-50 percent (Pauley and Bonar). Triploid grass carp are developed when eggs of a normal (diploid) pair of grass carp are shocked chemically, with excessive pressure, or with heat. Triploid progeny alleviated the major concern about grass carp, reproduction in the wild.

In 1983, WDFW and Ecology initiated a long-term agreement through the University of Washington, funded in part by the US Army Corps of Engineers, US Fish and Wildlife Service, and the US Environmental Protection Agency. The goal of the study was to determine if triploid grass carp could be used safely and effectively to control nuisance levels of aquatic plants in Washington. Results of the studies are summarized under impacts due to grass carp; further reading includes Thomas and Pauley 1987, Thomas et al. 1990a, Thomas et al. 1990b. In 1990, WDFW produced a policy for introduction of grass carp to Washington lakes, ponds, or reservoirs less than or greater than five acres but without public access, and lakes, ponds or reservoirs with public access.

Permits are most readily obtained if the lake or pond is privately owned, has no inlet or outlet, and is fairly small. The objective of using grass carp to control aquatic plant growth is to end up with a lake that has about 20 to 40 percent plant cover, not a lake devoid of plants. In practice, grass carp often fail to control the plants or all the submersed plants are eliminated from the waterbody. The Washington Department of Fish and Wildlife determines the appropriate stocking rate for each waterbody when they issue the grass carp stocking permit. Stocking rates for Washington lakes generally range from 9 to 25 eight- to eleven-inch fish per vegetated acre. This number will depend on the amount and type of plants in the lake as well as spring and summer water temperatures. To prevent stocked grass carp from migrating out of the lake and into streams and rivers, all inlets and outlets to the pond or lake must be screened. For this reason, residents on waterbodies that support a salmon or steelhead run are rarely allowed to stock grass carp into these systems.

Once grass carp are stocked in a lake, it may take from two to five years for them to control nuisance plants. Survival rates of the fish will vary depending on factors like presence of otters, birds of prey, or fish disease. A lake will probably need restocking about every ten years. Success with grass carp in Washington has been variable. Sometimes the same stocking rate results in no control, control, or even complete elimination of all underwater plants. It has become the consensus among researchers and aquatic plant managers around the country that grass carp are an all or nothing control option. They should be stocked only in waterbodies where complete elimination of all submersed plant species can be tolerated.

Fish stocked into Washington lakes must be certified disease free and sterile. Sterile fish, called triploids because they have an extra chromosome, are created when the fish eggs are subjected to a temperature or pressure shock. Fish are verified sterile by collecting and testing a blood sample. Triploid fish have slightly larger blood cells and can be differentiated from diploid (fertile) fish by this characteristic. Grass carp imported into Washington must be tested to ensure that they are sterile. Because Washington does

not allow fertile fish within the state, all grass carp are imported into Washington from out of state locations. Most grass carp farms are located in the southern United States where warmer weather allows for fast fish growth rates. Large shipments are transported in special trucks and small shipments arrive via air.

WDFW has the primary regulatory responsibility for stocking grass carp, however, other agencies have participated in or funded research on the use of grass carp for aquatic plant control and will continue to do so.

Grass carp effectively control some species of aquatic plants by feeding on them. The amount and rate of plant-biomass reduction is directly related to grass-carp feeding rates and the number of fish introduced (stocking rate). This feeding rate depends on several factors, including grass-carp age, water temperature, and dissolved oxygen level. Because grass carp prefer some species to others, the rate at which plant biomass is reduced also depends on the type of plants available for consumption.

Researchers at the University of Washington, who have been studying grass carp since 1983, do not recommend use of grass carp for Eurasian watermilfoil control. This species is not a preferred food source and grass carp will consume most other aquatic plants before eating this species. Generally Eurasian watermilfoil is consumed only when the waterbody is overstocked with grass carp and no other food source is left. This sometimes results in the total eradication of all submersed species in a waterbody. Grass carp should be stocked for Eurasian watermilfoil management only if total eradication of all submersed species can be tolerated.

The University of Washington has developed a stocking model designed to maintain 30% to 40% of aquatic vegetation in a lake, for use as a management tool by the WDFW. University researchers recognize that each system should be evaluated to determine if stocking rates will meet the variety of lake management goals in Washington (Thomas et. al. 1990). In practice Bonar et. al. found that only 18 percent of 98 Washington lakes stocked with grass carp at a median level of 24 fish per vegetated acre had macrophytes controlled to an intermediate level. In 39 percent of the lakes, all submersed plant species were eradicated.

Use of grass carp to control aquatic vegetation may result in adverse environmental impacts, with the potential for adverse impacts increasing if carp are stocked at inappropriate levels. Introduction of grass carp has been shown to reduce waterfowl abundance because grass carp and waterfowl prefer some of the same plant species and may compete with each other for sustenance. Because grass carp do not discriminate between target and non-target species, they may eliminate threatened or endangered plant species and/or alter wetland composition. Generally in Washington, grass carp do not consume emergent wetland vegetation or water lilies even when the waterbody is heavily stocked or over stocked. A heavy stocking rate of triploid grass carp in Chambers Lake in Thurston County resulted in the loss of most submersed species, whereas the fragrant water lilies, bog bean, and spatterdock remained at pre-stocking levels. A stocking of 83,000 triploid grass carp into Silver Lake, Washington resulted in the total eradication of all submersed species, including Eurasian watermilfoil and Brazilian elodea. However, extensive wetlands in Silver Lake have generally remained intact. In southern states, grass carp have been shown to consume some emergent vegetation.

Grass carp can live up to 20 years or more and are very difficult to capture. Once grass carp are stocked into a waterbody, they can only be removed with very great difficulty. A rotonone bait was recently registered which can remove about 1/3 of the grass carp population. Fish are trained to feed at a pellet feeder. Once fish are trained a rotonone impregnated pellet is substituted and any fish consuming the bait are killed. However, remaining grass carp will not eat the bait. Pauley and Bonar evaluated seven techniques as methods of capture for grass carp in five Washington lakes. The capture methods included angling, pop-

nets, lift nets, or traps in baited areas, angling in non-baited areas, heating the water in small areas to attract the fish, and herding fish into a concentration area and removing them with gill nets or seines. Herding fish into a concentrated area was the most effective technique when followed by angling in baited areas. As noted in the "methods" section, the WDFW has developed several conditions designed to mitigate some of the impacts identified above.

Toxicity Use of grass carp is not expected to release toxic materials.

2. Impacts Due to Grass Carp

Earth

Sediments Although European carp (a separate species) are known to increase the turbidity of water by disturbing sediments, grass carp do not pull up plants by the roots like the common carp but eat from the top down without disturbing roots or sediment. However in situations where grass carp have completely eliminated all submersed aquatic plants, grass carp will consume organic matter from the sediments, stirring them into the water column in the process. Removal of aquatic plants also allows wind mixing to suspend sediments into the water increasing total suspended solids and turbidity.

Removal of plants by carp grazing may decrease the sedimentation rate in lakes, while waste from carp may increase sedimentation. Increased waste may also facilitate nutrient recycling through algal populations.

Water

Surface Water Baseline data obtained by the University of Washington suggest that dense stands of aquatic macrophytes can have a significant effect on water quality in shallow lakes of the state (Pauley and Thomas 1987). The formation of a canopy can partition the water column into areas of contrasting water quality, with elevated pH, increased water temperature, and supersaturated dissolved oxygen concentrations within watermilfoil mats. Beneath the surface canopy, water circulation and light penetration are restricted, while temperature and dissolved oxygen are reduced.

Dense beds of macrophytes can potentially modify the internal loading of phosphorus in lakes as a result of physical-chemical changes beneath plant beds, especially decreased dissolved oxygen. Removal of large dense beds of macrophytes by grass carp grazing may affect sediment release of phosphorus.

Introduction of grass carp may reduce the aquatic plants from dense to moderate densities, which should improve water quality in part due to increased mixing of the water by wind. Total devegetation does impact water quality in Silver Lake where stocking grass carp resulted in total eradication of submersed vegetation, the benthic animal populations went from zero to a healthy community. This was attributed to increased wind mixing of the water column, which allowed oxygen to reach the formerly anoxic sediments. However, wind mixing also decreased water clarity by stirring sediments into the water column.

Bonar et. al. investigated the impacts of stocking grass carp on the water quality of 98 Washington lakes and ponds. They found that the average turbidity of sites where all submersed macrophytes were eradicated was higher (11 nephelometric turbidity units (NTU's) than sites where macrophytes were controlled to intermediate levels (4 NTU's) or not affected by grass carp grazing (5 NTU's). Most of this turbidity was abiotic and not algal. Chlorophyll a was not significantly different between levels of macrophyte control.

Introduction of triploid grass carp into Keevies Lake and Bull Lake in Washington resulted in a reduction of surface cover and biomass of the aquatic macrophytes along with some improvements in the water quality. In areas dominated by floating leaved species, mean bottom dissolved oxygen increased from < 1 mg/liter to > 3 mg/liter. Mean conductivity increased from around 30 to 90 useiemens, and was associated with higher

ion concentrations, primarily calcium which increased from around 2 mg/l to 4 mg/l. In areas dominated by submergent species, surface pH was reduced to <10, surface dissolved oxygen decreased from >20 mg/l to around 10-15 mg/l and mean bottom dissolved oxygen increased from 2.0 mg/l to 4.5 mg/l.

If aquatic plants are rapidly eliminated, the influx of nutrients from grass carp feces could result in substantial changes in water chemistry, phytoplankton densities (especially cyanobacteria, i.e., bluegreen algae), and bacteria levels (Pauley and Thomas 1987). Not sure this has been proven out in the field.

Water Chemistry Low concentrations of dissolved oxygen beneath plant canopies can in some cases lead to the release of phosphorus from the sediment into overlying water. The most important change in redox in natural, stratified sediment-water systems (where Fe^{+++} is most responsible for phosphorus fixation with O_2) happens in the redox (Eh) range of 3.8-3.1, which corresponds to the reduction of $\text{Fe}(\text{OH})_3$ to Fe^{++} . Consequently, phosphorus is released from the sediment into overlying water. Such low values have been observed below dense beds of aquatic vegetation in Washington lakes. (Detailed descriptions of dissolved oxygen changes with depth in Eastern and Western Washington lakes with and without grass carp can be found in Pauley and Thomas 1987, Thomas et al. 1990a, and Thomas et al. 1990b.)

Public Water Supplies Grass carp introduction would have no effect on public water supplies beyond effects described under Surface Water.

Plants

Habitat Grass carp have been used successfully to control certain species of aquatic plants around the world (Appendix F). They prefer some species of plants and will not consume others. Two types of aquatic plant control are desirable with grass carp in Washington:

1. Total and rapid eradication of plants where water flow and navigation are important (an example is an irrigation system where water delivery is more important than habitat), and
2. Slow reduction of plants to intermediate levels to enhance fish production and water dependent recreation.

Reaching the above goals will depend both on the stocking rate (number of fish added to the lake) and the knowledge of feeding preferences of grass carp on aquatic vegetation.

Pauley and Bonar performed experiments to evaluate the importance of 20 Pacific Northwest aquatic macrophyte species as food items for grass carp. Grass carp did not remove plants in a preferred species-by-species sequence in the multi-species plant communities. Instead they grazed simultaneously on palatable plants of similar preference before gradually switching to less preferred groups of plants. The relative preference of many plants was dependent upon what other plants were associated with them. The relative preference rank for the 20 aquatic plants tested was as follows: *Potamogeton crispus* = *P. pectinatus* > *P. zosteriformes* > *Chara* sp. = *Elodea canadensis* = Thin-leaved *Potamogeton* > *Egeria densa* (large fish only) > *P. praelongus* = *Vallisneria americana* > *Myriophyllum spicatum* > *Ceratophyllum demersum* > *Utricularia vulgaris* > *Polygonium amphibium* > *P. natans* > *P. amplifolius* > *Brasenia schreberi* = *Juncus* sp. > *Egeria densa* (fingerling fish) > *Nyphaea* sp > *Typha* sp. > *Nuphar* sp.. Researchers also demonstrated that feeding rates of triploid grass carp on four macrophyte species increased at higher water temperatures.

In field tests, investigators determined that many plant species less desirable to humans (such as *M. spicatum*, *E. canadensis*,) overwinter vegetatively and are able to grow significantly in spring when water is less than 18° C. Consequently, when the grass carp's body temperature rises enough to feed, it has to remove a large standing crop of the above macrophytes before it can control their regrowth (Pauley and Thomas 1987).

Plant species in lakes exhibit variability in growth patterns that effect the ability of grass carp to control them. For example, broadleaf communities tend to peak late in the growing season when ambient water temperatures are higher, which may help grass carp to control these species more effectively. In contrast,

the maximum biomass of filamentous submerged communities tends to occur earlier in the season before carp metabolism is sufficient to control it.

University of Washington researchers investigated effects of grass carp introduction on five Washington lakes, two west of the Cascades and three on the eastern side of the mountains (Thomas et al. 1990b). In western Washington lakes dominated by *Brasenia schreberi* and *Potamogeton natans*, declines in *P. natans* and *B. schreberi* increased after grass carp introduction, as did the total amount of open water. In the eastern Washington lakes which were dominated by *Elodea canadensis*, *P. pectinatus*, *Myriophyllum sibiricum*, and *Ceratophyllum demersum*, *P. pectinatus* was removed after grass carp stocking and the amount of open water increased in all sites. When stocked for lake management, grass carp usually show the most significant impact 3-5 years following introduction.

Bonar et al investigated the effects of grass carp on aquatic macrophyte communities and water quality of 98 Washington lakes and ponds stocked with grass carp between 1990-1995. Noticeable effects of grass carp on macrophyte communities did not take place in most waters until two years following stocking. After two years, submersed macrophytes were usually either completely eradicated (39 percent of the lakes), or not controlled (42 percent of the lakes). Control of submersed macrophytes to intermediate levels occurred in 18 percent of lakes at a median stocking rate of 24 fish per vegetated acre.

Ecology does not support removal of non-noxious emergent (wetland) species except in controlled situations where the intent is to improve low quality (Category IV) wetlands and in situations where wetlands have been created for other specific uses such as stormwater retention.

Grass carp eat native species as well as exotic species of aquatic vegetation; thus use of grass carp may result in positive or negative impacts depending on vegetation in the specific waterbody. Negative impacts could include invasion by less desirable species such. Another potential negative impact of grass carp introduction would be destruction of perimeter or riparian emergent vegetation. Loss of perimeter vegetation may increase shoreline erosion and decrease the treated waterbody's value as wildlife habitat.

Animals Grass carp are omnivorous in the juvenile stage and will eat small invertebrates once they are beyond the egg sac stage. When grass carp are larger than one inch they convert to herbivory. Since grass carp are stocked at sizes over 8 inches long, they are not expected to graze invertebrates in Washington lakes. Additionally, triploid grass carp are sterile, thus eliminating any chance of reproduction in the wild.

The greatest potential impact of grass carp introduction on invertebrates and vertebrates is the removal of the majority of the plant community. Major changes in aquatic vegetation will affect invertebrate populations that depend on it; however, no negative impacts to fish have been documented in studies in Washington (Appendix F). Under some circumstances, complete plant removal is detrimental to largemouth bass populations, but may be beneficial to salmonids. Populations of small centrarchid fish are generally considered to become more vulnerable to predation as aquatic macrophyte densities decrease, and populations of piscivorous centrarchid fish become highest at intermediate densities of aquatic plants (Wiley et al. 1984, in Thomas et al. 1990a). At extremely high densities of grass carp where aquatic macrophytes have been totally eradicated, growth and abundance of centrarchid gamefish populations have been poor (Thomas et al. 1990a).

Pauley et. al. studied the impacts of triploid grass carp grazing on the game fish assemblages of Pacific Northwest lakes. Fish samples were taken from Keevies Lake and East Pipeline Lake in Washington in 1885, 1986, 1988, and 1990, and from Devils Lake, Oregon in 1986, 1987, and 1988. Age, length, and weight data were collected for several species of fish including largemouth bass (*Micropterus salmoides*), black crappie (*Pomoxis nigromaculatus*), pumpkinseed sunfish (*Lepomis gibbosus*), bluegill sunfish (*Lepomis macrochirus*), yellow perch (*Perca flavescens*), and brown bullhead (*Ictalurus nebulosus*). In Devils Lake, largemouth bass, bluegill sunfish, and yellow perch exhibited post stocking declines after grass carp were introduced. East Pipeline Lake exhibited no effect on the largemouth bass subsequent to grass carp stocking. Keevies Lake exhibited declines of largemouth bass after grass carp were introduced. Pauley et. al. attributed the declines of bass and other fish in Devils Lake to increased angler access while the bass

declines in Keevies were thought to be due to natural variation. In neither case were grass carp thought to be responsible for any game fish population changes.

Although effects of plant removal on largemouth bass (*Micropterus salmoides*) and sunfish (*Lepomis* spp.) have been studied after introduction of grass carp, the relationship between macrophytes and these fish is poorly understood (Thomas et al. 1990a). It is unlikely that grass carp would physically disturb spawning bluegill sunfish by causing turbidity and siltation in spiny-ray spawning areas. It is also unlikely that grass carp stocked at sizes over 8 inches will be potential prey for largemouth bass. Indirectly, removal of aquatic macrophytes is assumed to increase susceptibility of most forage fish to game fish predation.

Grass carp have been diagnosed with over 100 diseases and parasites, 29 documented in the US. The top 11 pathogens are already present in Washington or are not considered dangerous, with the exception of the Asian tapeworm (*Bothriocephalus opsarichthydis*). Importation of the tapeworm will be avoided by shipping only grass carp that are greater in length than 8 inches (Appendix E).

Decreased food availability for waterfowl is another potentially negative impact of grass carp introduction which may change the quantity and quality of available aquatic plant food for waterfowl (Appendix F). Grass carp and some waterfowl prefer similar plants (Hardin et al. 1984, in Thomas et al. 1990a). In other locations in the US, declines in waterfowl abundance have been observed after grass carp grazing.

Grass carp are riverine fish and have the urge to move into flowing water. Therefore all inlets or outlets need to be screened to keep grass carp from migrating up or down stream. Screening in a waterbody with anadromous fish runs is problematic. It is difficult and expensive to design a screen that will allow salmon or steelhead passage while restricting the movement of grass carp. Grass carp grow to be large athletic fish fully capable of negotiating fish ladders. In fact, in 1996 presumably escaped grass carp were observed migrating past several lower Columbia and Snake River dams (Loch and Bonar). Generally WDFW will not allow the stocking of grass carp into systems that support anadromous fish runs. However, there have been exceptions such as Silver Lake in Cowlitz County.

Threatened and Endangered Species. Introduction of grass carp has the potential to affect submersed and emersed plant species federally listed as rare, threatened or endangered. Applications for grass carp stocking for each specific site should include a review of the rare, threatened, or endangered plant species listed by US Fish and Wildlife and of "proposed sensitive" plants and animals listed by Washington State Natural Heritage Data System.

Water, Land and Shoreline Use

Aesthetics Use of grass carp may result in decreased vegetation, which may be viewed as either a positive or adverse impact on aesthetics, depending on the attitude of the observer, and the amount and species of plant removed.

Recreation When stocked at proper rates into lakes with dense macrophyte beds, grass carp will improve swimming, fishing, and boating. If stocked at too high a rate, grass carp could potentially decrease fish habitat and thus negatively affect fishing. Negative impacts on aquatic vegetation used by waterfowl are expected; decreased waterfowl populations would negatively affect hunting. Grazing by grass carp is expected to improve recreational facilities used for swimming, fishing, and boating by decreasing unwanted aquatic vegetation.

Navigation Effects of grass carp on transportation are expected to be minor. Grazing of dense macrophyte beds by grass carp may improve navigation, most likely for recreational boating.

Agriculture No impacts on agricultural crops are expected with grass carp introductions. Grass carp are currently used successfully in irrigation canals in California, Arizona, and Alberta. At this time, grass carp are proposed for use in manmade irrigation and power canals in Washington at the expense of the property owner.

3. Mitigation, Grass Carp

Communications. For lakes, ponds, or reservoirs less than or greater than five acres and without public access, triploid grass carp may be planted at the expense of the property owner. A list of all property owners with land adjacent to the water and their opinion of the proposed introduction must be provided to WDFW. Lakes, ponds, or reservoirs with public access may be stocked with grass carp if a professional lake restoration feasibility assessment or an integrated aquatic vegetation management plan is completed. Both types of planning efforts must include public input and involvement (Appendix E).

Permits WDFW requires a game fish planting permit before allowing grass carp into a pond or lake. Ecology can fund some grass carp projects through Phase II Lake Restoration Grants or loans, or by the Aquatic Weeds Management Fund if grass carp are identified as a management option in an integrated aquatic plant management plan for that waterbody. If inlets or outlets require screening prior to the introduction of grass carp, a HPA also needs to be obtained from WDFW for the screening work. . Department of Natural Resources Natural Heritage Program must be contacted for assessment of threatened and endangered species before WDFW will permit the stocking of grass carp.

Water Quality, Plants, and Animals. Impacts of grass carp on water quality, plants, and animals are continuing to be assessed. Potential impacts from grass carp include changes to water chemistry, increased phytoplankton densities resulting from an influx of grass carp feces, and loss of desirable or unique plant species and/or excessive loss of plant biomass. Because waterfowl depend on aquatic plants for food, loss of plant biomass may adversely effect waterfowl. Information regarding impacts to fish populations and wetlands is equivocal and warrants additional research. Monitoring of a recent stocking of grass carp into Silver Lake will help us understand potential impacts to emergent vegetation. As the lead permitting agency for stocking grass carp, WDFW has developed policies designed to reduce or prevent some potential impacts. A copy of this policy and other relevant information is included in Appendix E.

WDFW requires documentation from the US Fish and Wildlife Service that fish to be planted are certified disease-free triploid grass carp. A professional lake restoration feasibility assessment must be conducted to address cultural resources, water quality, restoration feasibility, and public involvement as well as a SEPA checklist for all applications requesting permission to stock grass carp. In evaluating each of these checklists, WDFW can assess potential impacts to specific water bodies and condition permits to reduce potential impacts. Because most permits issued to date have been for small, privately owned lakes with impacts identified as being minimal, the responsible official has determined that DNSs were appropriate. Where shoreline permits, or other local permits are required, local government may be the lead agency.

Impacts from grass carp depend on characteristics of the waterbody to be stocked, the stocking rate, the plant community, plant density, and the knowledge of feeding preferences of grass carp. WDFW generally permits the introduction of grass carp mostly into small, private ponds. However, their policy does not contain a waterbody size threshold and the agency has received permit applications for larger waterbodies. WDFW's policy states that Ecology must approve applications to waterbodies with public access, which may affect the number of applications to larger systems.

Limiting permits to small, privately-owned ponds tends to reduce the scope of impacts, as well as the seriousness of impacts such as potential cumulative effects on wildlife, particularly waterfowl. Impacts may be reduced by assessing habitat needs, surveying existing habitat in a general area, evaluating potential cumulative impacts of habitat reduction in waterbodies in that area, and preserving habitat adequate to meet the needs of waterfowl.

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Section VII. Alternative 5: Use of Chemicals Only

A. Introduction to Chemical Control Methods

Under Alternative 5, Ecology would process water quality modifications to allow appropriate use of any aquatic herbicide registered for use that meets the criteria included in Washington State regulations and standards that would not cause unreasonable adverse impacts, including prolonged use restrictions. This is primarily the “No Action” alternative where we maintain current practices, the difference being the addition of newly assessed chemicals.

This Section updates the “Use of Chemicals Only” sections of the 1980 Aquatic Plant Management Environmental Impact Statement and its 1992 Supplement and adds new data on 2,4-D and formulations of Aquathol and Hydrothol 191. This section will be further updated spring 2001 to include risk assessment information on diquat, triclopyr and copper compounds registered for aquatic use in Washington State. The current sections on diquat and copper have not been updated but will be when the risk assessments for those herbicides are completed. Triclopyr was not included in the original or supplemental EIS, so it will be a new addition to the 2001 SEIS.

Since new risk assessments are not planned for fluridone and glyphosate, changes in application practices and labeling since the 1992 SEIS have been noted and are reflected in this supplement where appropriate. Health and ecological risk are considered in light of the previous EIS documents and any new data or labeling related to the products in question.

The information on each herbicide reviewed in this Draft SEIS is brief, concise and not overly technical. Analysis and evaluation of the recently assessed compounds is based primarily on technical review found in the risk assessments supporting them and is simply summarized herein. Where the Draft SEIS contains general information and conclusions, the detailed technical supporting information is referenced in the respective risk assessment appendix.

In response to requests by members of the Steering and Technical Advisory Committees, and as allowed by SEPA Rule WAC 197-11-430, the impact and mitigation information for each herbicide has been combined, rather than set in separate sections, to make this document more accessible for general guidance and reference. For purposes of uniformity, the herbicides reviewed in the 1992 SEIS that will not be updated at this time have been reorganized into the same format. However, the complete bibliographies for those herbicides are included at the end of the respective sections.

The supportive risk assessments found in the appendices follow the structural organization that the Environmental Protection Agency (EPA) Office of Pesticide Programs uses to develop data requirements for the registration of pesticides. They include basic data on the physical and chemical properties of each herbicide, the behavior of the compounds in the environment, and their toxicity to non-target organisms. These data contribute to the quantification of hazard. The suite of data developed in this manner have been evaluated under the use scenarios (the labeled directions for use) in order to determine exposure. Then, the risk assessment process combines the hazard and exposure data to determine the magnitude, if any, of risks for the use of the products when used according to the label. Where risks are identified, seasonal timing, rate or use limitations, or other criteria are suggested as possible risk mitigation criteria.

The environmental and human health review of each compound is comprised of five sections:

- The registration status of the compounds under review,
- The physical and chemical characteristics of the herbicide's active ingredients, and where relevant, the characteristics of the end use products,
- A review of potential environmental and human health impacts from exposure to the use of the compounds. This section combines the assessment of the effect data with the behavioral properties of the compounds in order to quantify risk for non-target organisms. Fate and hazard data are also combined in a formal risk assessment for the compounds under review,
- The final part quantifies hazard or risk for the use of the products when used according to the label and proposes mitigation measures for each aquatic herbicide. Where risks are identified, seasonal timing, rate or use limitations, or other criteria are suggested as possible risk mitigation criteria,
- And either a reference to the supporting appendix or a complete bibliography of citations is presented at the end of each herbicide review.

B. Types of Herbicides

Herbicides are selected for use based on toxicity, availability, cost and effectiveness of control. Effectiveness of an aquatic herbicide is primarily dependent on its mode of action and suitability for the targeted aquatic plant. Aquatic plants are categorized as submerged, emergent or floating, indicating the way the plant typically grows. Plants growing only below the water line are submerged, those growing from below the water line to above the waterline are emergent, and those growing on the surface of the water, sometimes un-rooted, are floating. Pre-emergent and post-emergent weed control refers to whether control measures are taken prior to or after germination of first growth of the plant. Herbicides used for aquatic weed control fall into one or more general categories:

- Contact herbicides are plant control agents that are used in direct contact with foliage and destroy only contacted portions of the plant.
- Systemic herbicides are applied to foliage and/or stems and are translocated to roots or other portions of the plant, resulting in death of the entire plant.
- Broad-spectrum herbicides kill most if not all plants with the appropriate dosage.
- Broadleaf herbicides generally kill dicot plants with broad leaves.

C. Registration Requirements

In order to register a pesticide for use with the EPA, the active ingredient and its formulations must be tested for mammalian toxicity, physical chemistry, environmental fate, effects on ground water, and ecotoxic effects. Additional work must be done to demonstrate expected magnitude of residue on edible products and residues in water. After these data are generated, they are submitted to EPA's various branches for review. If the reviews find that the product does not pose significant risk to man, livestock, or wildlife and has a favorable environmental persistence and degradation profile, a registration will be granted. With that registration the manufacturer has permission to sell the product in the United States. However, each state may have its own separate registration process which may be more stringent than the EPA's registration process. Washington State's registration procedure follows EPA registration. It requires that the applicant submit a copy of EPA registration label and a copy of the confidential statement of formula. Washington State Department of Agriculture reviews these submittals for compliance with state and Federal requirements. If these requirements are filled the product will usually

be registered unless it presents an unusual hazard to the environment. A more detailed description of the registration process is given in Appendix B, the Introduction to SEIS Assessments of Aquatic Herbicides and in the registration status sections of each herbicide appendix.

D. Tank Mixes, Inerts and Surfactants

In general, tank mixes are not permitted in the state of Washington for the control of aquatic weeds in public waterways. Occasionally, endothall will be mixed with copper sulfate for the control of algae in impounded golf course ponds. It is believed that this combination has better algae controlling properties than either of the compounds alone (Appendix D, Vol. 2, Sect. 4, p. 36).

Not all formulations have similar toxicity on an acid equivalent basis. “Inert materials” in a formulation may interact with the pesticide to give antagonistic, additive, cumulative or synergistic effects against target plants (aquatic weeds and algae) and non-target fish and aquatic invertebrates. For example, endothall acid is considerably more toxic to rainbow trout and bluegill sunfish when certain “inerts” are added, possibly due to a synergistic effect (Appendix D, Vol. 2, Sect. 4, p. 36).

If surfactants are used, care should be taken to use ones registered for aquatic uses since they have potential toxicity to fish. Thickening agents like Polysar® or Nalquatic® are used in other states to control drift with liquid endothall products that are applied to floating weeds and may also allow subsurface applications to sink more deeply into the water column where they can be most effective. However, these two surfactants are not registered for use in Washington State and therefore are not allowed for use here (Appendix D: Sect. 4, p. 36 and Personal Communication with Wendy Sue Bishop, WSDA, May 30, 2000).

E. General Mitigation for Aquatic Herbicides

Introduction Several strategies are available for avoiding or minimizing potential impacts associated with use of aquatic herbicides. Some mitigation measures can be applied generally to all proposed herbicide treatments because there are impacts common among various treatments. Some mitigation measures must be tailored to each specific proposal and/or chemical proposed for use. General mitigation measures that may be incorporated into all permits allowing the use of aquatic herbicides are discussed below. This general mitigation section is supplemented by a discussion of measures designed to reduce potential impacts of specific aquatic herbicides in each herbicide section. It will be further updated once mitigation sections for the herbicides under current review are finalized.

General Mitigation The mitigation conditions listed below are used as general conditions in Ecology’s Short-Term Modification Order (the permit boilerplate) dated March 2000. A few minor changes to the standard “boilerplate” language regarding notification have been made based on Ecology-wide consensus during a meeting of the permit writers and headquarters staff (Ecology HQ, May 23, 2000). The changes generally add flexibility to timing requirements regarding notification without compromising intent of requirements. These conditions are routinely used in conjunction with other relevant materials and considerations by agency officials when making final decisions on permit conditions.

General Mitigation Conditions:

G-1 The applicator shall comply with all pesticide (including herbicide and adjuvants) label instructions. When application conditions issued by Ecology differ from those on pesticide labels, the more stringent of the two requirements must be used. No condition shall reduce the requirements on the pesticide label.

G-2 All persons applying pesticides should be aware of the following regulations:

- A. The pesticide applicator regulations as required by the Washington Department of Agriculture (RCW 17.21, RCW 15.58, and WAC 16-228).
- B. Public access policy and Hydraulics Code regulations as required by the Washington Department of Fish & Wildlife (RCW 75.20.100, WAC 220-110).
- C. Shorelines regulations as required by the local city or county (RCW 90.58).
- D. All applicable regulations of other agencies. Check local ordinances for compliance.

G-3 A. The applicator shall furnish a list of planned treatments to Ecology's appropriate regional office by noon one (1) working day prior to the day of treatment. This list shall contain the following information:

- The names of the waterbodies (as written in Specific Conditions: S1 of this Order) that are planned for treatment, in the order that they are planned for treatment;
- An estimate of the hour the application will begin;
- The location on the waterbody where treatment will begin; and
- The pesticide(s) expected to be used.

In the event there is a schedule change of more than one-half (½) hour, then the applicator shall notify the appropriate Ecology regional office of the new starting time of treatment at least two (2) hours prior to the time of beginning any treatment. A message by voice mail or FAX shall suffice for this condition.

B. The applicator shall notify the appropriate Ecology regional office by the close of the day of the scheduled treatment (12:00 midnight) by an answering device or by FAX, the following:

1. Reasonable estimate of time, location on the waterbody and pesticide(s) applied; or
2. Proposed date, location on the waterbody and pesticide of a cancelled pesticide application.

G-4 The applicator shall notify the Department of Agriculture's (WSDA) Pesticide Management Division at voice - (360) 902-2040 or FAX - (360) 902-2093 for treatments west of the Cascade Mountains or voice - (509) 575-2746 or FAX - (509) 575-2210 for Yakima; or voice - (509) 625-5229 or FAX (509) 625-5232 for Spokane; or voice - (509) 664-3171 or FAX (509) 6643170 for Wenatchee - for treatments east of the Cascade Mountains by the close of the previous business day before applying pesticides to any waterbody. This notice shall include a reasonable estimate of the time of day the application is expected to take place, the location on the waterbody where treatment is expected to begin, and the pesticide(s) expected to be used.

G-5 In the event of an unauthorized discharge (spill) of chemicals, gasoline, oil or other contaminants into state waters, or onto land with a potential for entry into state waters, containment and cleanup efforts shall begin immediately and completed as soon as possible, taking precedence over normal work. Cleanup shall include legal disposal of any spilled material and used cleanup material.

The applicator shall also immediately call the twenty-four (24) hour number of the appropriate regional office.

G-6 The applicator shall immediately call the twenty-four- (24) hour number of the appropriate Ecology regional office if the applicator learns of any person who exhibits or indicates any toxic and/or allergic response, or of any fish, fauna, or non-targeted plants that exhibit stress conditions or die following a pesticide treatment.

G-7 The applicator shall not cause recreational water use restrictions (i.e., restrictions on swimming or fish consumption) to occur during Memorial Day weekend, July 4th weekend, Labor Day weekend, or the opening day of any applicable fishing season. The applicator shall also minimize the occurrence of water use restrictions during non-holiday weekends. Non-holiday weekend treatments that will require water use restrictions must be for emergency situations only and will require written approval by Ecology.

G-8 A. The applicator shall keep complete application records on the approved spray report form. This form will also fulfill the WSDA's reporting requirements.

B. These application records shall be completed and available to Ecology the same day the herbicide(s) were applied and be mailed or hand delivered to Ecology immediately upon request.

G-9 During all pesticide applications, the applicator, or persons applying pesticides, shall possess, on-site, the authorizing Order for the application of herbicides.

Public Notice Procedures:

P-1 Residential and Business Notice Procedures:

A. The applicator shall complete copies of the Herbicide Application - Residential and Business Notice form. These forms shall be sent to all residences and businesses within one-quarter ($\frac{1}{4}$) mile in each direction along the shore and five hundred-(500) feet upland of the areas to be treated. No later than the day following distribution of the Herbicide Application - Residential and Business Notice, a copy and the date of distribution of the notice shall be mailed or faxed to the Ecology contact (identified in G-3 A).

B. Notification shall take place ten (10) to twenty-one (21) days prior to initial treatment. When planning copper treatments for algae, if less than thirty (30) days remain between the date of the issuance of the permit and the date planned for initial treatment, notification shall take place one (1) to twenty-one (21) days prior to initial treatment.

C. If the Herbicide Application - Residential and Business Notice explains the application schedule for the whole season, and there is no significant deviation from that plan, no further Herbicide Application - Residential and Business Notice [as required under P-1 (A)] will be required for the rest of the season (unless a resident or business specifically requests further notification).

If the location(s) to be treated change by over one hundred (100) feet, or the date of the treatment extends five (5) days before or after the dates set for treatment, or different pesticide(s) are proposed for use, another Herbicide Application - Residential and Business Notice shall be issued. The use of copper compounds to control algae shall be exempt from this requirement.

D. The one-quarter (1/4) mile zone of notification discussed in P-1 (A) shall be expanded for the use of glyphosate (Rodeo®). In this case, the applicator shall notify all residences and businesses within one-half (1/2) mile in each direction along the shore and five hundred (500) feet upland of the treatment area.

E. Distribution of the Herbicide Application - Residential and Business Notice may be done by mail to residences or businesses, or by handbills given directly to the residences or businesses. If handbills are used, the applicator shall secure the notices to the residences or businesses doorknob in a fashion that will hold them in place but will not damage property. If the residence or business is gated or guarded by watch dogs, the applicator may secure the notice in clear view on the outside of the gateway or may attach the notice to the outside of the residence in a fashion that will hold it in place but will not damage property.

A copy of the notice and a list of names and addresses where they were sent shall be kept by the applicator for seven (7) years and be hand delivered or mailed to Ecology immediately upon request.

F. When using fluridone (Sonar®) and/or glyphosate (Rodeo®), the applicator shall include a statement in the Herbicide Application - Residential and Business Notice informing residents and businesses of the one-quarter (1/4) mile and one-half (1/2) mile application restriction for potable water use [i.e., water treated with glyphosate (Rodeo®) should not be used for drinking water within one-half (1/2) mile of the treatment site].

G. Conditions in P-1 (A-D) shall not apply to waterbodies that are entirely owned by the sponsor and occupied only by them and their immediate family, have no public access, and have no inlet or outlet.

If all the residents within the standard notification area [one-quarter (1/4) mile in each direction along the shore and five hundred (500) feet upland] are part of a homeowner's association, and no public access exists to the waterbody, the public notice conditions in P-1(A-D) may be waived if all residents are informed through a homeowner's association newsletter or similar mailing thirty (30) days prior to treatment.

A copy of the newsletter and its mailing list shall be kept by the applicator for seven (7) years and be hand delivered or mailed to Ecology immediately upon request.

P-2 Legal Notice Procedures:

The applicator shall publish a notice in the legal notices section of a local newspaper of general circulation (or nearest regional paper if a local paper does not exist) for all pesticide applications expected during the time the permit is in effect.

These legal notices shall be published ten (10) to twenty-one (21) days prior to the first pesticide application of the season. This notice shall include:

- A. The pesticide(s) to be used and their active ingredient(s);
- B. The approximate date(s) of treatment;
- C. The approximate location(s) to be treated;
- D. Any water use restrictions or precautions;
- E. The posting procedure; and

- F. The names and phone numbers of the applicator and the appropriate Ecology regional office.

A dated copy of the published notice or an affidavit from the legal department of the newspaper shall be kept by the applicator for seven (7) years and be hand delivered or mailed to Ecology immediately upon request.

P-3 Posting Procedures:

The applicator shall post all signs prior to the application of any pesticide(s), but no more than twenty-four (24) hours prior to application. The applicator shall use good faith and reasonable effort to ensure that posted signs remain in place until the end of the period of water use restrictions or forty-eight (48) hours for Rodeo and copper. The applicator shall be responsible for removal of all signs before the following treatment of the waterbody or before the end of the treatment season, whichever comes first.

The applicator shall construct and post signs as follows:

A. Small signs shall be copied from the templates in Appendix "C" of this order. For larger, two- (2) by-three- (3) foot templates for posting at public access sites, contact the appropriate regional office.

B. Posting Shoreline Private Property Areas:

Signs shall be a minimum of eight and one-half (8½) by eleven (11) inches in size and be made of a durable weather-resistant material. Lettering shall be in bold black type with the word "WARNING" (or "CAUTION") at least one- (1) inch high and all other words at least a one-quarter- (¼) inch high.

Sign board color for the first seasonal treatment of a waterbody shall be white, for the next treatment the signboard color shall be yellow, and the following treatment the sign board color shall be orange. The sign board color for the fourth treatment shall be white, the fifth yellow, the sixth orange, etc.

Signs must face both the water and the shore and be placed on each private property within ten (10) feet of the shoreline adjacent to the treatment area(s). Where a private property shoreline is greater than one hundred-fifty (150) feet, the applicator shall post one sign for every one hundred (100) feet of shoreline. Signs shall be posted so they are secure from the normal effects of weather and water currents, but cause no damage to private or public property.

When using pesticides with swimming and/or fish consumption restrictions or precautions, the applicator shall extend the zone of shoreline posting to include all property within four hundred (400) feet of the treatment area(s). When copper compounds are used, no private shoreline posting is required.

C. Posting Shoreline Public Access Areas:

Public access areas include: swim beaches, docks, and boat launches at resorts; privately-owned community access areas; and public access areas.

Signs shall be a minimum of two (2) feet by three (3) feet in size and be made of a durable weather-resistant material. Lettering shall be in bold black type with the word "WARNING" (or "CAUTION") at least two (2) inches high and all other words at least a one-half- (½) inch high. The colors used for the sign board shall be white, yellow, or orange.

Signs must face both the water and the shore and be placed within twenty-five (25) feet of the shoreline. Where the public access has a shoreline length greater than one hundred-fifty (150) feet, the applicator shall post one sign for every one hundred (100) feet of shoreline. The applicator shall place signs so they are clearly readable by people using the access areas. Signs shall be posted so they are secure from the normal effects of weather and water currents, but cause no damage to private or public property.

An eight and one-half- (8½) by-eleven- (11) inch weather resistant map detailing the treatment areas for each herbicide used shall be attached to the sign. The map shall identify the location(s) of the pesticide(s) used and mark the reader's location at the public access site.

These public notice signs shall be posted at all of the waterbody's public access areas within one-quarter (¼) mile of the treatment area and all of the waterbody's public boat launches within one and one-half (1.5) miles of the treatment area. NOTE: When using pesticides with swimming and/or fish consumption restrictions or precautions, the applicator's map shall include a four hundred- (400) foot buffer strip around the treatment area(s).

D. Posting on the Water:

When the pesticide to be used does not have swimming and/or fish consumption restrictions or precautions, posting buoys on the water is not necessary. When the waterbody is less than one acre and/or less than two hundred (200) feet from the treatment area to the opposite shore, posting by buoys is not necessary. When the entire shoreline is restricted by one treatment, no buoys shall be required.

The applicator shall use buoys to mark treatment area boundaries on the water. Durable weather-resistant signs are to be attached to a buoy so they are readable from two opposing directions. The applicator shall position signs so they are completely out of the water. The signs must be at least eight and one-half- (8½) by-eleven (11) inches in size. Lettering shall be in bold black type and the word "WARNING" (or "CAUTION") shall be at least one- (1) inch high and all other words shall be at least a one-quarter- (¼) inch high. The colors used for the sign board shall be white, yellow, or orange.

The applicator shall space buoys so there is one at each approximate corner of the treatment area and at one hundred- (100) foot intervals around the treatment area. Treatment areas of one hundred- (100) foot diameter or less shall be marked with one buoy in the center of the treatment or at one hundred- (100) foot intervals around the treatment area. The applicator shall place buoys so they form a fifty- (50) foot buffer strip around the treatment area(s).

P-4 For areas where tank mixes of different chemicals are applied to the same water column, the applicator shall adhere to the posting and notification requirements for the pesticide with the most extensive or stringent requirements or precautions.

P-5 When the EPA label or Ecology Order restricts human consumption of fish, any posted signs or other forms of notification shall explicitly state that restriction. Do not state or imply the lake is closed to fishing unless the Department of Fish & Wildlife has closed the lake.

P-6 Warning signs shall be posted in English and the language commonly spoken by the community who use the area.

P-7 The applicator shall obtain advance written approval from the appropriate Ecology regional office before making variations to the posting and notification procedures. Refer to Condition G-3 for regional telephone numbers.

F. Sediment Mitigation for Aquatic Herbicides

Sediments have only just begun to be researched and regulated as an environmental resource, as explained in the following excerpt from *Bioassessment Analysis of Steilacoom Lake Sediments*:

The assessment of adverse effects of contaminated sediment on fish and invertebrate populations exists as a major problem for aquatic toxicologists. Contaminant material generally precipitates, forms various complexes or adsorbs and binds to particulate matter (Giesy et al., 1990). Ultimately, sediment serves as the final repository for the pollutant. Benthic organisms can be directly impacted via the ingestion of particulate matter or continual re-exposure due to leaching and re-suspension of contaminant material resulting from physical disturbances to the sediment (Geisy et al., 1988). Bio-availability of sediment contaminants depends on many factors, including physical properties of the sediment and the contaminant and physical and biological properties of overlying water. Water quality criteria are based on the concentration of a particular substance in solution in the water column. Sediment criteria have only recently begun to be established. (Henry, M.G., Morse, S. and Jaschke, D. 1991. Minnesota Cooperative Fish and Wildlife Research Unit.)

The anti-degradation and designated use policies of the Sediment Management Standards (Chapter 173-204-120 WAC) state, in part, that existing beneficial uses must be maintained and that sediment must not be degraded to the point of becoming injurious to beneficial uses. Additionally, sediment in waters considered outstanding natural resources must not be degraded; outstanding waters include those of national and state parks and scenic and recreation areas, wildlife refuges, and waters of exceptional recreational or ecological significance. The purpose of the standards is to manage pollutant discharges and sediment quality to protect beneficial uses and move towards attaining designated beneficial uses as specified in section 101(a)(2) of the Federal Clean Water Act (33 USC 1251, et. seq.) and Chapter 173-201A WAC, the State's surface water standards.

The sediment standards include specific marine-sediment chemical criteria, but the criteria for low salinity and freshwater sediments have not yet been developed. Sections of Ecology's Sediment Standards have been reserved in anticipation of development of criteria for these sediment environments.

G. 2, 4-D Aquatic Herbicide Formulations

1. Registration Status

Three active ingredients of 2,4-D have been approved by WSDA; however, only one of these active ingredients (2,4-D butoxyethyl ester) has been approved by Ecology for aquatic weed control. Both the EPA and Washington State have approved two labeled products, Aqua-Kleen® (distributed by Nufarm) and Navigate® (distributed by Applied Biochemists) for control of aquatic macrophytes in lakes and

ponds. Aqua-Kleen® and Navigate® are identical products sold under different labels (Appendix C, Vol. 3, Sect. 1, p.3).

2. Description

2,4-D (2,4-Dichlorophenoxy acetic acid) is the active component in a variety of herbicide products used for both terrestrial and aquatic application sites. 2,4-D is a selective plant hormone type product that is translocated within the plant to the susceptible sites. Its mode of action is primarily as a stimulant of plant stem elongation. 2,4-D stimulates nucleic acid and protein synthesis and affects enzyme activity, respiration, and cell division. It is absorbed by plant leaves, stems, and roots and moves throughout the plant, accumulating in growing tips. Its primary use is as a post-emergent herbicide.

2,4-D is formulated in a multitude of forms, however only two active ingredient forms are currently being supported by the manufacturers for use in aquatic sites. These are the dimethylamine salt and the butoxyethyl ester. The butoxyethyl ester (BEE) is the active ingredient in the two products used in Washington State. Because these are products for use in aquatic sites, the physical chemical characteristics and data reported are limited to the pure acid active ingredient (and technical product), the dimethylamine salt and the butoxyethyl ester. The majority of the data was obtained from a 1996 Food and Agricultural Organization (FAO) document. This document was extensively peer reviewed for the purposes of chemical and physical properties and was found to be relatively complete and up to date (Appendix C, Vol. 3, Sect. 2, p.3).

Typical Use Granular 2,4-D butoxyethyl ester (Aqua-Kleen® and Navigate®) is a post-emergent systemic herbicide used primarily to control watermilfoil and water stargrass. The other 2,4-D product used around aquatic sites is 2,4-D Dimethylamine salt (2,4-D DMA). This product is primarily used for control of water hyacinth and brush control along ditchbanks. Another 2,4-D product registered in Washington for the control of noxious weeds is 2,4-D 2-Ethylhexyl ester (2,4-D 2-EHE). 2,4-D 2-EHE is not registered for control of aquatic weeds but is typically used to control purple loosestrife (*Lythrum salicaria*) and brush along ditchbanks. Species other than those listed on the labels may be controlled fully or in part by application of these products; however, the distributor makes no efficacy claims for control of weed species not listed on the label (Appendix C, Vol. 3, Sect. 1, p.8).

Risk Analysis Treatment of commercial fish ponds with 2,4-D sodium salt (a surrogate of 2,4-D acid) produces no direct impact on the biota of these ponds. Secondary effects have been seen that produce increases in heterotrophic bacterial count, phytoplankton count, zooplankton count, benthic invertebrate biomass and subsequently benthic feeding fish survivability and yield (biomass). In general, similar effects were observed with chronic risk quotients as well. Chronic risk quotients generally predict the chronic safety of these herbicides to fish, free-swimming invertebrates and sediment invertebrates. While chronic risk quotients generally predict chronic safety to fish, free-swimming invertebrates and benthic invertebrates when they are in the water column, accurate prediction of chronic safety or lack of safety from exposure to treated sediment is not possible without an understanding of bioavailability factors that could mitigate the toxic effects of 2,4-D BEE sorbed to sediment (Appendix C, Vol. 3, Sect. 4, pp. 108-109).

Data Gaps The role of sediment in removing 2,4-D from the environment should be investigated along with the effects of 2,4-D in sediment on benthic organisms. Levels of granular 2,4-D BEE in the sediment is particularly important since, under some circumstances, it is known to accumulate to very high levels. Whether or not these high sediment levels are biologically available to benthic species that reside on the surface or in the interstitial areas of the sediment has not been adequately evaluated.

Furthermore, the effects of digestion on those species that consume sediment to extract nutrients are unknown. Due to the extreme sensitivity of certain benthic organisms to 2,4-DMA and 2,4-D BEE our risk assessment leads us to conclude that although 2,4-D products were safe to most organisms (90 to 95%), the most sensitive benthic organisms may not be protected. Therefore, the toxicity of 2,4-D in overlying water, interstitial water and whole sediments needs further investigation.

Further investigations need to be conducted to determine which levels of 2,4-D are safe to sensitive, threatened and endangered species (particularly Chinook salmon and sea-run trout). Additional studies, including sea-water challenge tests emphasizing species indigenous to the Northwest should be conducted so that risk due to exposure can be managed more effectively. This is of particular concern for benthic organisms since regulators, registrants, the applicator community and the general public have recently expressed great concern over this issue (Appendix C, Vol. 3, Sect. 4, p. 111).

3. Environmental and Human Health Impacts

Earth

Soil Half-lives of 2,4-D acid in soil generally ranged from 2 days to about 12 days at 17-25°C, with 2,4-D from granular applications being on the higher end of that range. A half-life of 39 days was reported for 2,4-D acid when a forest was treated with the DMA salt of 2,4-D. Reduction of soil moisture to about 50% of capacity or less increased half-lives, in some cases dramatically. Temperature was shown to be a factor in the length of persistence, with lower temperatures increasing half-lives. In one study, acid half-lives were much longer in soils taken from 2 to 4 foot depths compared with those from the top foot of soil. These results illustrated the contribution to increased persistence of sparser soil microorganism populations and less organic carbon in the lower depths.

A major metabolite of 2,4-D in soil is CO₂. Substantial amounts of soil humic and fulvic acids have been reported as metabolic products in soil studies, as have traces of 2,4-dichlorophenol and 2,4-dichloroanisole. In pure culture metabolism tests of 2,4-D acid using soil microorganisms, numerous other related compounds have been seen. Much of the carbon in the 2,4-D molecule is taken up by soil microorganisms and used to build cell tissues or used in their metabolic processes like carbon from any other source. The small amounts of numerous compounds seen are likely intermediate compounds caught in a "snapshot" of the metabolic process (Appendix C, Vol. 3, Sect. 3, pp.10-11).

Sediment In sediment, the BEE formulation may persist from a few weeks to as long as 3 months, with occasional instances of persistence to 6-9 months, though the latter is unusual. Longer sediment persistence is probably facilitated by the use of granular formulations that release acid over a prolonged period. If the BEE or acid is in contact with flocculent (light, fluffy) sediment, adsorption to the sediment particles and subsequent slow release may prolong the presence of residues near and in the sediment (Appendix C, Vol. 3, Sect. 3, p.18).

Erosion Since these products are not generally applied terrestrially, classical erosion effects typically do not occur. However, removal of plants from irrigation canal situations may result in erosive processes occurring to a limited extent (Appendix C, Vol. 3, Sect. 3, p.20).

Air

Inhalation Toxicity Acute inhalation overexposure to 2,4-D in animal studies have demonstrated signs of respiratory tract irritation, e.g. salivation, lacrimation, mucoid nasal discharge, labored breathing, dried red or brown material around the eyes and nose. None of the signs persisted beyond 3-7 days post exposure, nor were there any deaths (FAO, 1996). No signs of systemic toxicity following 2,4-D exposure have been reported (Appendix C, Vol. 3, Sect. 5, p.19).

Drift 2,4-D is normally applied as a granule (2,4-D BEE) or at or below the water surface (2,4-D DMA); thus accidental "drift" exposure to upland vegetation during application would be minimal with the exception of emergent aquatic plant communities bordering the treated area. If any proposed "sensitive" plants or candidate species under review for possible inclusion in the state list of endangered or threatened species occurs along the banks of waterways to be treated with 2,4-D products, the applicator should leave a protective buffer zone between the treated area and the species of concern (Appendix C, Vol. 3, Sect. 4, p.61).

Water

Surface Water Breakdown of 2,4-D by hydrolysis in sterile water is pH dependent. The overall pattern is that 2,4-D is rapidly broken down in natural pond and lake systems in a few days, while the resulting acid is usually below detection levels (approximately 0.01 ppm) in treated area water within a month.

BEE breaks down to 2,4-D acid in aquatic systems. The major degradates of the acid are 2,4-dichlorophenol (immediate) and CO₂ (final). Humic and fulvic acids bound to the sediment are also important degradates. Small amounts of dichloroanisole, 4-chlorophenol, and related compounds have also been reported. Much of the carbon in the 2,4-D molecule is taken up by soil microorganisms and used to build cell tissues or used in their metabolic processes like carbon from any other source. As is the case for soil, the minor products are likely intermediate compounds caught in a "snapshot" of the metabolic process (Appendix C, Vol. 3, Sect. 3, pp.18-19).

Photolysis Only one report of photolysis was found. In that study, no significant breakdown of BEE in sterile pH 5 water was observed at up to 30 days of light exposure. (Photolysis of BEE vapor in air was found to occur with a half-life of 13-20 days.) Photolytic degradation of 2,4-D acid was found at pH 3.5, 6.8, and 8.9 with a half-life of about 70 minutes. However, another study at pH 6 found no significant degradation of 2,4-D acid after 8 hours. The major product of BEE photolysis is 2,4-D acid. When the acid is photolyzed, the primary product is probably 2,4-dichlorophenol, which breaks down further under light to smaller amounts various intermediates, with the final products appearing to be humic acids (Appendix C, Vol. 3, Sect. 3, p.7).

Groundwater Over the many years of its use as a terrestrial herbicide, 2,4-D has been detected in wells and other groundwater samples. Washington State (1993) quotes Dynamac (1988) in reporting 2,4-D detection in about 100 of more than 1700 groundwater samples from nine states, but 2,4-D has not generally been found to contaminate groundwater. The most likely routes for contamination are spills during mixing of application solutions at wellheads, illegal dumping, surface water runoff from treated fields, and movement down through the soils from heavily treated agricultural land. With respect to groundwater movement, the difference between terrestrial uses of 2,4-D and aquatic weed control uses is that lakes provide, in essence, an isolated incubator in which 2,4-D degradation can take place without immediate impact on surrounding soil (Appendix C, Vol. 3, Sect. 3, p.47).

In some situations, 2,4-D has been seen in ground water where recharge areas have been treated with 2,4-D BEE. These recharge areas usually had porous bottoms (sand or gravel) with clay layers located below the bottom of the well shaft. Most down stream water treatment plants will not experience concentrations of 2,4-D higher than the Federal drinking water standard (0.07 mg/L) due to extensive dilution and lateral mixing (Appendix C, Vol. 3, Sect. 4, p.36).

Water Chemistry Exposure of living plant tissue to 2,4-D products or other herbicides usually results in secondary effects that may impact the biota. When plants start to die, there is often a drop in the dissolved oxygen content associated with the decay of the dead and dying plant material. Reduction in dissolved oxygen concentration may result in aquatic animal mortality or a shift in dominant forms to those more tolerant of anaerobic conditions. There may also be changes in the levels of plant nutrients due to release of phosphate from the decaying plant tissue and anoxic hypolimnion. Also ammonia may be produced from the decay of dead and dying plant tissue which may reach levels toxic to the resident biota. Ammonia may be further oxidized to nitrite (which is also toxic to fish), and the almost nontoxic, nitrate. The presence of these nutrients may cause an alga bloom to occur. If significant living plant biomass persists after treatment, the released nutrients may be removed before an algal bloom can occur (Appendix C, Vol. 3, Sect. 3, p.30).

Public Water Supplies The Aqua-Kleen label warns that this product is not to be applied to waters used for irrigation, agricultural sprays, watering dairy animals or domestic water supplies. However, risk assessment results of chronic exposure assessments indicate that human health should not be adversely impacted from chronic 2,4-D exposure via ingestion of fish, ingestion of surface water, incidental ingestion of sediments, dermal contact with sediments, or dermal contact with water (Appendix C, Vol. 3, Sect. 5, p.46).

Wetlands The presence of 2,4-D products at concentrations effective against plants in wetland environments may adversely effect wetlands. Dilution effects should mitigate the effects of 2,4-D so that it does not affect aquatic plants or non-target animals in marshes, bank and estuarine areas. The presence of 2,4-D in the lotic (moving water) environment, due to outflow from a lake or pond, may cause the destruction of aquatic plants that are favorable to the production of appropriate habitat for sunfish, minnows and bass. The subsequent habitat, with a low level of aquatic weed cover and a benthos consisting primarily of sand and gravel, would be more appropriate to the production of salmonids.

The estuarine environment may be affected by the use of 2,4-D. Some of the most susceptible species of invertebrates are estuarine species including grass shrimp, glass shrimp, and seed shrimp. The estuarine crab (*Uca uruguayensis*) has been eliminated from some estuarine areas due to the effects of 2,4-D. It is unclear if this is due to an avoidance response or an acute or chronic toxicity response. The presence of estuarine crabs and estuarine shrimp like those mentioned above are critical since they are important to the maintenance of the food web that attracts both game fish and fish of commercial importance. Anaerobic sediment typically found in estuaries can lead to the production of 2,4-chlorophenol or 4-chlorophenol which are both very toxic to some species of aquatic macrophytes, marine phaeophytes, various beneficial fungal species and amphipods. Failure to control emerged, floating, marginal and bank exotic (non-native) plants can cause the native vegetation to be crowded out, producing dense monoculture stands of noxious and invasive weeds, leading to the degradation of natural habitats and an economic burden for residents who must keep water flowing or navigable (Appendix C, Vol. 3, Sect. 4, p. 102).

Plants

Algae For the most part, 2,4-D products are not toxic to indicator species of algae, particularly 2,4-D DMA and 2,4-D acid. An exception may be freshwater and saltwater diatoms. 2,4-D products have a low toxicity to most blue-green algae at higher concentrations. There is some evidence that alga numbers increase when a water body is treated with 2,4-D DMA or 2,4-D acid for the control of Eurasian watermilfoil due to the release of nitrogen and phosphate. The phytoplankton cell count may double within a few days or weeks of treatment with 2,4-D. There may also be shifts in dominant species to those which find water temperatures and nutrient concentrations that occur after milfoil lysis ideal for growth (Appendix C, Vol. 3, Sect. 4, p. 5).

Food Chain 2,4-D BEE has a tendency to accumulate in sediment and plants from 1-7 days (Gangstad, 1986). This may be a reflection of plants and sediments “metabolizing” 2,4-D to products that can be incorporated into the plant structure or the sediment (as humus). Animals, however, rapidly hydrolyze adsorbed 2,4-D BEE to 2,4-D acid and excrete it unchanged back into the water. 2,4-D should not bioaccumulate; it should be rapidly eliminated from any organisms that ingest it; and it should not bioaccumulate as it passes up the food chain.

Eurasian watermilfoil apparently bioconcentrates 2,4-D to levels 33 to 94-fold higher than the levels found in water, but eliminates this material within 16-weeks after the watermilfoil mass has undergone extensive decay. The release of 2,4-D from decaying watermilfoil probably has little effect on the concentration of 2,4-D in water since the highest concentration in plants is only about one percent of the total 2,4-D in the aquatic system (Appendix C, Vol. 3, Sect. 4, p. 30).

Animals

2,4-D BEE will have no significant impact on the animal biota acutely or chronically when applied at rates recommended on the label. Although laboratory data indicate that 2,4-D BEE may be toxic to fish, free-swimming invertebrates and benthic invertebrates, data also indicate that toxic potential is not realized under typical concentrations and conditions found in the field. This lack of field toxicity is likely due to the low solubility of 2,4-D BEE and its rapid hydrolysis to the practically non-toxic 2,4-D acid (Appendix C, Vol. 3, Sect. 4, p. 11).

Bioconcentration in plants and animals is not likely for 2,4-D DMA, 2,4-D BEE or their hydrolysis/dissociation product (2,4-D acid). Although short term bioaccumulation of 2,4-D BEE can be fairly high in fish, after three hours of exposure 2,4-D BEE is converted to 2,4-D acid and excreted. If fish are “fed” 2,4-D acid, >90% is excreted within 24 hours. Work conducted in the field tends to corroborate this data since it was found that fish have little tendency to bioconcentrate 2,4-D in the field and when it does bioconcentrate it is rapidly eliminated.

One experiment showed benthic organisms and free-swimming invertebrates bioaccumulate 2,4-D to very high levels in the field, these results are probably artifacts since this experiment was not carried out to equilibrium. However, since these high levels are not found in fish, 2,4-D apparently does not bioconcentrate or biomagnify across trophic levels (Appendix C, Vol. 3, Sect. 4, p. 30).

Habitat Sites that have never been exposed to 2,4-D products may degrade 2,4-D DMA, 2,4-D BEE and 2,4-D acid more slowly than sites that have a previous exposure history. It may take several weeks for bacteria capable of using 2,4-D as their sole carbon source to develop out of the lag-phase and rapidly degrade applied 2,4-D DMA or 2,4-D BEE. Such rapid degradation leads to a half-life in ponds and rice

paddies of 1.5 to 6.5 days. However, if degradation, sorption and dilution factors are interacting in open waterways, the field dissipation half-life may be even shorter. Typical half-lives in Northwest waters are less than one week. Therefore, long-term persistence of 2,4-D BEE at concentrations that will cause environmental damage is not likely. Furthermore, since 2,4-D BEE has a low solubility and is rapidly hydrolyzed to the generally less toxic 2,4-D acid, the likelihood of 2,4-D BEE coming into significant contact with sensitive members of the biota is much reduced.

Initial elimination of exotic plants should increase habitat for fish (Bain & Boltz, 1992). Growth and reproduction of fish may be more due to general metabolic stimulation of benthic microorganisms and subsequent greater availability of fish food stock than a precise control of the amount of habitat available (Appendix C, Vol. 3, Sect. 4, p. 61).

Fish 2,4-D BEE, has a high laboratory acute toxicity to fish (rainbow trout fry and fathead minnow fingerlings). Formal risk assessment indicates that short term exposure to 2,4-D BEE should cause adverse impact to fish. 2,4-D acid has a toxicity similar to 2,4-D DMA to fish for the common carp and rainbow trout, respectively.

Limited field data with sentinel organisms (caged fish) and net capture population surveys indicate that 2,4-D BEE lacks acute environmental toxicity to fish when applied at labeled rates. Exposure of smolts of several salmon species to 1 mg/L 2,4-D BEE for 24-hours did not affect the ability of these smolts to survive a subsequent 24-hour seawater challenge. This indicates that 2,4-D BEE probably does not interfere with the parr to smolt metamorphosis in anadromous fish species (Appendix C, Vol. 3, Sect. 4, p. 64). Although bluegill sunfish and rainbow trout bioaccumulate 2,4-D BEE for the first 3 hours of exposure to 1 mg 2,4-D BEE/L, the material is rapidly metabolized to 2,4-D acid and eliminated from the tissues in the next 48 to 120 hours. Several species of fish including sheepshead minnow and mosquito fish are known to avoid 2,4-D BEE at concentrations typically found in the field. 2,4-D BEE and 2,4-D acid produce a number of behavioral effects, pathological and metabolic effects at concentrations that are much higher than those typically encountered in the field. These effects are typical signs of stress in fish.

2,4-D BEE is moderately toxic to free-swimming daphnids and highly toxic to moderately toxic to most benthic invertebrates. Since the risk quotient is higher than the acute level of concern of 0.1 for benthic invertebrates, this segment of the biota is potentially at risk from the acute effects of 2,4-D BEE. However, the low solubility of 2,4-D BEE and rapid hydrolysis to 2,4-D acid would tend to limit exposure to the much less toxic 2,4-D acid. 2,4-D acid has a toxicity similar to the low toxicity of 2,4-D DMA to most species of invertebrates. For free-swimming invertebrates, the toxicity of 2,4-D acid and its sodium salt leads to a toxicity evaluation of practically non-toxic for these species. The level of concern is also not exceeded for the most sensitive species of benthic invertebrate (Appendix C, Vol. 3, Sect. 4, p. 9).

Amphibians Acute data for 2,4-D DMA salt and 2,4-D acid are available for several species of amphibians (the frog *Limodynastes peroni*, Indian toad *Bufo melanostictus* and the Leopard frog). The data indicate that 2,4-D DMA is relatively non-toxic to amphibians while 2,4-D acid is relatively non-toxic to the Leopard frog and moderately toxic to the Indian toad. Although these data are limited to only a few studies it appears that 2,4-D acid may be more toxic in these species than 2,4-D DMA (Appendix C, Vol. 3, Sect. 4, p. 99).

Birds Acute oral data for 2,4-D acid and 2,4-D BEE are available for several different species of birds (See Table 30: Appendix C, Vol. 3, Sect. 4, p. 193). These data indicate that the 2,4-D acid is moderately

toxic to practically nontoxic to birds when orally dosed and that 2,4-D BEE is practically nontoxic to birds when orally dosed (Appendix C, Vol. 3, Sect. 4, p. 99).

Mammals Acute oral data are available for more than one mammalian species. These data indicate that 2,4-D BEE is slightly toxic and that 2,4-D acid is moderately to slightly toxic. Subchronic and chronic effects indicate that terrestrial species may be affected by long term exposure to 2,4-D acid in the diet (Appendix C, Vol. 3, Sect. 4, p. 100).

Water, Land and Shoreline Use

Agriculture At typical use rate concentrations, irrigation or flooding of crops with water that has been treated with 2,4-D DMA damages some crops and non-target wild plants. Although early growth stage damage has been observed on many crops including sugar beets, soybeans, sweet corn, dwarf corn and cotton, no significant reductions in yield were seen at harvest for most crops. Residue levels that would interfere with the marketability of crops were not seen in various crops including potatoes, grain sorghum, Romaine lettuce, onions, sugar beets, soybeans, sweet corn or dwarf corn. 2,4-D will not bioaccumulate in crop plants or fish at levels that will interfere with their marketability or consumption (Appendix C, Vol. 3, Sect. 4, p. 43).

Pastureland flooded with water containing 2,4-D may lead to the destruction of various turf plants. In addition, sensitive crop plants like Mexican red beans, lentils, peas, grapes and tomatoes may be irreversibly damaged by the presence of 2,4-D in irrigation or floodwater. Other non-target plants may be adversely impacted (Appendix C, Vol. 3, Sect. 4, pp. 102 and 154).

Reentry and Swimming Use of the chemical in accordance with label directions is not expected to result in adverse health effects. Results indicate that 2,4-D should present little or no risk to the public from acute exposures via dermal contact with sediment, dermal contact with water, or ingestion of fish. A review of the acute, subchronic and chronic toxicology investigations demonstrate that 2,4-D acid, amine salts and esters have similar degrees of low systemic toxicity. The amine salts and esters are metabolized to the acid and undergo rapid excretion by the kidneys. 2,4-D does not accumulate in the organism or environment; however, when the administered dose exceeds the threshold for normal renal function, a decrease in excretion occurs resulting in possible systemic poisoning. Findings from subchronic and chronic toxicology studies and genotoxicity testing do not implicate 2,4-D as a carcinogen or developmental or reproductive toxin in laboratory animals. A review of the epidemiology studies and opinions from scientific review panels indicate that some of the investigations present inconsistent results, design flaws, and contain confounding variables. Therefore, based on the weight-of-the-evidence, label directed use of 2,4-D for aquatic herbicide control poses little concern for causing adverse health effects to people (Appendix C, Vol. 3, Sect. 5, p. 46).

Dermal contact with vegetation may present limited risk one hour after application. By 24 hours, post-application non-carcinogenic risk is essentially nonexistent, as 2,4-D is unavailable for dermal uptake. Margins of safety for all acute exposure scenarios are greater than "100", implying that risk of systemic, teratogenic (causing fetal malformations), or reproductive effects to humans is negligible. Results of chronic exposure assessments indicate that human health should not be adversely impacted from chronic 2,4-D exposure via ingestion of fish, ingestion of surface water, incidental ingestion of sediments, dermal contact with sediments, or dermal contact with water, including swimming (Appendix C, Vol. 3, Sect. 5, p. 46)

4. Mitigation

Use restrictions 2,4-D is not an algicide. Use only according to label for macrophytes. Aquatic formulations of 2,4-D have not been evaluated for aerial applications in Washington State. Aqua-Kleen® and Navigate® applied at concentrations of 100-lbs.formulation/acre will control Eurasian watermilfoil and spare most species of native aquatic vegetation. Although removal rates of Eurasian watermilfoil can approach 95%, when eradication is the goal, treatment up to two times per year may be necessary (Appendix C, Vol. 3, Sect. 4, pp.119-120).

Swimming/skiing All General Mitigation posting requirements apply. Informational buoys should be placed around the treatment area with an enforced 24-hour swimming restriction. Swimming outside the treatment area is permitted.

Boating No special restrictions recommended for boaters entering the area of treatment.

Drinking/Domestic Uses According to current labels (03/99) aquatic herbicide formulations of 2,4-D may not be applied to waters used for irrigation, agricultural sprays, watering dairy animals, or domestic water supplies. Labels (03/99) expressly forbid use in or near a greenhouse.

Fisheries 2,4-D BEE, has a high laboratory acute toxicity to fish (rainbow trout fry and fathead minnow fingerlings). Formal risk assessment indicates that short term exposure to 2,4-D BEE should cause adverse impact to fish. However, during actual field applications of 2,4-D granules, fish show little impact. This is probably due to the insolubility of the BEE formulation in water. It is considered highly probable that fish are actually being exposed to the acid formulation of 2,4-D which is considered practically non-toxic to fish. Follow label restrictions for oxygen ratios.

Endangered Species Based on warning information on Aqua-Kleen® and Navigate® labels for fish, extra caution should be taken when dealing with endangered species allowing for a level of concern of <0.05 rather than the more typical value of <0.1 for non-endangered species. Restrictions on seasonal applications are warranted to protect Chinook smolts from the effects of 2,4-D products.

Fish Consumption Results of chronic exposure assessments indicate that human health should not be adversely impacted from chronic 2,4-D exposure via ingestion of fish.

Wetlands Lakes with outflows to estuaries should be treated only at low flows or with a buffer around the outlet.

Post-treatment Monitoring If treatment occurs near domestic water sources, post treatment monitoring may be desirable.

References

Please refer to citations in Appendix C: Compliance Services International, 2000. *Supplemental Environmental Impact Statement Assessments of Aquatic Herbicides*, Volume 3: 2,4-D. 442 pages.

H. Copper Compounds

1. Registration Status

Copper was reviewed in the 1980 EIS, then updated with a more thorough review in the 1992 SEIS in response to uncertainties regarding copper's impact on aquatic systems. Given the known toxicity of copper compounds to aquatic life, primarily fish, and given the recent ESA listings of several salmonid species in Washington State waters, Ecology's Water Quality Program made a policy decision to disallow the use of copper in salmon-bearing waters in March, 2000. This decision affects all waters of the state used by salmonids and is currently going through the administrative process to become a formal, written policy. Copper will be assessed again in 2001.

There are currently nine products containing copper that may be used for control of algae and aquatic weeds in Washington State. They are copper sulfate distributed by Phelps Dodge (algicide), Captain® (elemental copper – liquid formulation) manufactured by Sepro (algicide), Nautique® manufactured by Sepro (Herbicide and Algaecide), Cutrine® Plus manufactured by Applied Biochemists (algicide), Cutrine® Granular manufactured by Applied Biochemists (algicide), K-Tea® manufactured by Griffen (algicide) and Komeen® manufactured by Griffen (herbicide), Cleargate® manufactured by Applied Biochemists (algicide) and Earthtec® (algicide) (Appendix D: Sect. 4, p. 69).

2. Description

Copper compounds are primarily used for algae control in Washington State. Algae are an integral part of healthy aquatic ecosystems, and are an essential food source to fish and other aquatic animals. However, deleterious algae blooms can occur in waterbodies with excessive nutrients. Dense algae blooms can adversely affect water quality, causing changes in water chemistry such as reduced dissolved oxygen and certain types of algae can be harmful to human health. While copper effectively controls algae and improves water quality in the short term, long-term control is not normally achieved with copper treatments.

Copper compounds for aquatic use are manufactured either as copper sulfate (pentahydrate) or as a copper chelate product. Both forms contain metallic copper as the active ingredient, but in the chelate forms, copper is combined with other compounds to help prevent the loss of active copper from the water. Copper complexes are principally formulated for aquatic plant and algae control and act as cell toxicants (Westerdahl and Getsinger 1988). The active ingredient listed in these formulations is usually copper as copper sulfate pentahydrate or copper as elemental (in ethanolamine, triethanolamine, and ethylenediamine copper complexes) (See Appendix G, p.1 for an in-depth technical description).

Copper sulfate is probably the most widely used chemical for the control of planktonic algae; its use as an algicide was first advocated in the United States by Moore and Kellerman (1904). Copper sulfate is selective in its algal toxicity, due to the formation of insoluble copper complexes under certain conditions (Maloney and Palmer 1956). Generally, copper sulfate does induce reduction in primary production, but effects are short term because copper concentrations in the water column return to pretreatment levels within a few days. An important factor controlling copper concentration in particulate materials is uptake by planktonic organisms. The kinds and amounts of dissolved organic material in the water are also important. Humic substances make up a large percentage of the dissolved organic material in fresh water and include refractory organic molecules. These substances may scavenge copper ions and thus play a major role in its transformation. (See Appendix G, p.2 for an in-depth technical information.)

Liquid formulations are applied using a hand or power sprayer or may be injected below the water surface (Westerdahl and Getsinger 1988). Copper compounds are not subject to photolysis or volatilization. Once copper has been used for aquatic macrophyte control, it persists indefinitely due to its elemental nature. EPA has established a 1 mg/l drinking water standard for copper.

3. Environmental and Human Health Impacts

Earth

Soils Use of copper compounds to control algae may result in increased water clarity. Increased clarity can lead to increased plant growth. Greater densities of plant vegetation can reduce current speed in flowing water that may in turn increase siltation. In general, indirect impacts to soils or topography should be slight with the aquatic use of copper compounds. (See following section on Sediments.)

Sediments The ultimate sink for copper in the aquatic environment is deposition in sediments, which form a reservoir of copper in freshwater environments. High concentrations of copper in sediments have been reported near some industrial sources, such as discharge zones of some power stations. Factors reported to affect the quantity of copper in sediments include the organic carbon content of the sediment and water, particle size distribution, pH, and copper concentration in the water. These factors may account for the considerable variability in copper content among samples collected under different circumstances. The effect of organic matter on the binding of metal ions does not seem to be simple (Harrison 1986). Furthermore, increases in copper concentrations are correlated with decreasing particle size.

Numerous studies support the notion that retention of copper in sediment is strongly influenced by the presence of organic material (See review in Chu et al. 1978). Organic material may be bound to the surface of particulate material and from this site acts upon the metal (Murray 1973). Walter et al. (1974) determined the occurrence of copper and other trace elements in lake sediment cores and found significant enrichment for most metals, including copper, within the upper 30 cm of sediment. They speculated that the principal factors for this enrichment phenomenon were oxidation-reduction reactions resulting from decay of organic material under anaerobic conditions and induced biochemical reactions in microbes under stress. Other experiments demonstrated that heavy metals in sediments showed upward migration resulting from bacterial mechanisms. Thus, even with continual sedimentation, copper is likely to remain concentrated in the upper strata of sediments (Chu et al. 1978).

Residence time, which is defined as the length of time required for all of the element to be removed and replaced by materials of other origins, has been estimated as 500,000 years for copper (Horne 1969).

Air

Adverse impacts to air quality are expected to be minor, such as a small amount of exhaust emissions associated with the use of application equipment. No adverse effects from aerial drift or overspray are expected since copper sulfate and copper complexes are not volatile.

Water

Surface Water Copper compounds are highly water soluble. However, once copper has been applied for alga or plant control, it persists indefinitely due to its elemental nature. The major processes affecting the

persistence of copper in aquatic systems are sediment sorption and physical export from the system. Both processes reduce the amount of copper in the aqueous phase; however sorption does not remove copper from the system. Copper has only been removed from the aqueous phase to the sediment phase and may remain in the system indefinitely.

A short-term effect of copper sulfate on surface water quality in some Minnesota lakes included dissolved oxygen depletion by decomposition of dead algae (Hanson and Stefan 1984). Repeated copper sulfate treatments also accelerated phosphorus recycling from the lake bed.

Ground Water No ground water contamination issue is associated with the use of copper compounds as aquatic algicides. There are no label restrictions against drinking, swimming, or fishing in waters treated with copper, but there is a 1 mg/l drinking water standard for copper. (See Appendix G, p.4 for an in-depth technical information.)

Public Water Supplies Trace amounts of copper are essential to human life and health. Like all heavy metals, copper is also potentially toxic. Physiological mechanisms have evolved to control the absorption and excretion of copper, which operate to offset the effects of temporary deficiency or excess of the metal in the diet. EPA has set 1.0 mg/l copper as criteria for domestic water supplies.

Only very large amounts of orally ingested copper are toxic. For example, acidic foods or beverages which have been in contact for a long time with copper metal may cause acute gastrointestinal disturbances. When copper enters the body following inhalation, absorption from burned skin, or absorption from a contraceptive device in the uterine cavity, toxicosis may result from amounts of copper that would not cause a problem when eaten.

EPA's Office of Pesticide Programs does not have laboratory toxicological data meeting their standards, therefore, they consider available information from literature sources. They report that "Oral ingestion of copper compounds is irritating to the gastric mucosa and emesis [vomiting] occurs promptly, thereby reducing the amount of copper available for absorption into the body. Only a small percentage of copper ingested is absorbed, and most of the absorbed copper is excreted." EPA is requiring additional human-health related data for only a few copper products.

Information provided by EPA, Office of Pesticide Programs is supplemented by a document prepared for EPA, Office of Drinking Water entitled, Review of the Drinking Water Criteria Document for Copper. The Science Advisory Board found reasonable a health-based drinking water standard of one mg/L (milligram per liter). Where recommended label rates are below 1 mg/L because scientists who reviewed the proposed standard found relevant the possibility of an increased sensitivity of 13 percent of the black population with G6PD deficiency.

Among the unknowns of copper formulations are "inert" ingredients. We do not know what inerts are used in various copper compounds and most inerts used in pesticides have not been tested to determine health and environmental effects. Inert ingredients constitute a major portion (as much as 92%) of many herbicides with copper as the active ingredient.

Plants

Habitat Copper has been widely used as an algaecide and herbicide for nuisance aquatic plants. It is known as an inhibitor of photosynthesis and plant growth; however, toxicity data on individual species are

not numerous. Copper appears to affect basic physiological processes such as growth and nitrogen fixation as well as photosynthesis and can produce distinct morphological changes in algae.

The optimal concentration range for essential trace elements in aquatic environments may be very narrow. Copper inhibits photosynthesis and growth of sensitive alga species at concentrations often found in pristine waters (as low as 1-2 ug/l total Cu).

The effect of pH on the toxicity of copper to algae can be important. Peterson et al. (1984) demonstrated that changes in metal toxicity with pH resulted from competition between H^+ and Cu^{+2} for cellular binding sites at the lower pH range, but at higher pH, copper was still toxic because of the decreased competition by H^+ . H^+ affects toxicity directly by competing with free metal ions for cellular uptake sites and indirectly by determining the chemical speciation of copper (i.e., the size of the free metal pool).

The response of primary producers to copper is dependent on species, life stage, and most importantly, the chemical form of copper in the water (Harrison 1986). Recovery of the alga population was observed within 7 to 21 days of copper sulfate treatment of several lakes in Minnesota (Hanson and Stefan 1984). Copper releases can have both direct and indirect effects on food-chain organisms because algae concentrate copper to a high degree. Direct effects result when the overall productivity of an ecosystem is reduced because decreased quantities of the primary producers are available for consumption by higher food-chain organisms. Indirect effects result when algae concentrate copper to high concentrations and are consumed by higher trophic levels, resulting in sublethal or lethal effects on sensitive species. (See Appendix G, pp.6-7 for additional technical information.)

Animals

Freshwater Invertebrates In general, the sensitivity of invertebrates to acute copper exposure is highly variable (Chu et al. 1978). Acute toxicity data (48- to 96-hr LC_{50} or EC_{50}) of copper for certain phyla used as freshwater test organisms show a wide range of results. Concentrations for arthropoda (crustaceans) ranged from 5 to 3000 ug/l, for annelida ranged from 6 to 900 ug/l, and mollusca ranged from 40 to 9000 ug/l (Leland and Kuwabara 1985). Harrison (1986) reports that acute toxicity LC_{50} values for crustaceans ranged from <10 to 9000 ug/l and for mollusca ranged from 39 to 2600 ug/l.

The largest amount of information available for any one group of crustaceans is on the acute toxicity of copper to daphnids. Daphnids are used as test organisms because they are a major component of freshwater zooplankton, are easily cultured, and are sensitive to contaminants. The same Daphnia species has demonstrated considerable differences in response to copper in numerous studies, perhaps due to experimental factors such as differing diet, water chemistry, species age, and loading density.

Chronic/ Sublethal studies of the effects of chronic exposures of invertebrates to copper are limited (Chu et al. 1978). Biesinger and Christensen (1972) observed a 3-week LC_{50} of 0.044 mg/l for Daphnia magna, and a 50% loss of reproduction at 0.035 mg/l. A concentration less than 0.035 mg/l was the highest continuous concentration that did not significantly decrease survival, growth, and reproduction. Winner and Farrell (1976) exposed four species of Daphnia to copper in the laboratory using a static method, water with 100-119 mg/l alkalinity, 130-160 mg/l hardness, 8.2-9. mg/l oxygen. The four species of Daphnia had decreased survivorship when exposed to 0.040 mg/l copper.

Vertebrates Trace metal toxicity to aquatic organisms is manifested in a wide range of effects, from slight reductions in growth rate to death. Occasional fish kills and a shift from game fish to rough fish may occur. Large differences are seen in the sensitivity of different species of fishes to copper. Acute toxicity (48-to

96-hr LC₅₀ or EC₅₀) data for copper for freshwater fish range from 10-900 ug/l for salmonidae, 700-110,000 ug/l for centrachidae, and 20-2000 ug/l for cyprinidae (Leland and Kuwabara 1985).

Fish It appears that the cupric ion is the chemical species that is toxic to fish, although CuOH⁺ might also be involved (Pagenkopf et al. 1974). pH is an important factor in determining cupric ion activity and hence copper toxicity (Chapman 1977). This relationship suggests that the acute lethal level of copper for a given species of fish for a given pH corresponds to cupric ion activity rather than total copper concentrations. A number of studies have demonstrated that copper toxicity is related to concentrations of about 0.040 mg/l are reported to be toxic to salmonid eggs, fry, fingerlings, juveniles, and adults (Chu et al. 1978). As expected, fish tested in water harder than 20 mg/l were less sensitive to copper, with copper toxicity roughly inversely proportional to water hardness. In general, cold-water species such as salmonids are more sensitive to copper than warm-water species (Chu et al. 1978). Most toxicity studies on salmonids have been performed with early life-stages ranging from eggs to juveniles while fewer studies have been performed to determine the relative sensitivity of older life stages.

Response to copper is not only dependent on species but on stage of development and sex. As fish develop, they undergo weight changes that affect their response to copper. Sac fry and early juveniles of eight freshwater fish were more sensitive than embryos to continuous exposures to copper (McKim et al. 1978). However, developing fish embryos are particularly sensitive to heavy metals during early embryogenesis. Permeability of the egg decreases and the chorion hardens during the first few hours after release, allowing the egg to become more resistant with time (Lee and Gerking 1980).

Shaw and Brown (1970) observed that rainbow trout eggs (*Oncorhynchus mykiss*, formerly *Salmo gairdneri*) could hatch following fertilization in a solution of copper (1000 mg/l), but hatching rate was significantly lower than unexposed controls. Grande (1967) demonstrated a reduction in egg hatching with copper exposure for rainbow, brown (*Salmo trutta*), and Atlantic salmon (*Salmo salar*). Brown trout were slightly more tolerant than the other two species. Copper inhibited egg development at the same concentration that was toxic to fry. However, concentrations as low as 0.02 mg/l had a sublethal effect (anorexia).

Hazel and Meith (1970) also concluded that Chinook salmon (*Oncorhynchus tshawytscha*) eggs were more resistant to copper than fry (acute toxicity to fry was observed at 0.04 mg/l, with inhibition of growth and increased mortality at 0.02 mg/l). McKim and Benoit (1971) observed that 0.185 mg/l had no effect on hatchability of brook trout eggs (*Salvelinus fontinalis*), but the same concentration drastically reduced survival and growth of alevin-juveniles. Thus, eggs appear to be more resistant to copper than other early stages.

Chapman (1977) tested the relative resistance of various life stages of Chinook salmon and steelhead trout (*O. mykiss*) to a number of metals and found that steelhead were consistently more sensitive than Chinook. Newly hatched alevins in both species were less resistant to copper than later juvenile stages.

In a study on effects of copper on adaptation of coho salmon (*O. kisutch*) from freshwater to seawater, ATPase activity, and downstream migration, Lorz and McPherson (1976) showed that yearling coho salmon exposed to ionic copper for 144 hours exhibited depressed ATPase activity and decreased survival in seawater in proportion to copper concentration (range: 0 to 0.080 mg/l). The sensitivity of juvenile fish to copper increased from November to May (of the following year) corresponding to the smolting period. Increased sensitivity to copper in May is related to onset of parr-smolt transformation. Smolts that are exposed to copper in freshwater often cannot survive in saltwater (copper concentration of 0.020 mg/l for

144 hours). Adult salmonids appear to be just as susceptible to copper as juveniles of the same species are (Chapman 1977).

Death in fish from copper acute toxicity may be due to the disruption of the respiratory process caused by damage to gill epithelium. Furthermore, copper may have a profound effect on hormone activity in salmonids; studies by Donaldson and Dye (1974) indicate that yearling sockeye salmon (*O. nerka*) exhibit a marked corticosteroid stress response when exposed to potentially lethal and sublethal concentrations of copper.

Holland et al. (1960) studied effects of copper sulfate and copper nitrate on Chinook salmon, where 50 percent mortality was observed between 42 and 96 hours at concentrations of 0.178 to 0.318 mg/l. Total kills occurred in 18 hours when fish were exposed to 1.00 mg/l copper and in less than 42 hours at 0.563 mg/l. At 0.563 mg/l, pink salmon (*O. gorbuscha*) showed significant mortality and loss of equilibrium. The minimum and maximum critical levels for coho salmon were 0.16 mg/l and 0.38 mg/l copper, respectively.

Sprague (1964) tested the toxicity of copper in soft water on Atlantic salmon. An incipient lethal level of 0.050 mg/l was estimated below which fish could survive indefinitely.

The 48-hour LC₅₀ for rainbow trout was 0.8 mg/l copper (Herbert et al. 1965). Brown (1968) also estimated 48-hour LC₅₀ values for the same species and reported a range of 0.4 to 0.5 mg/l. Trout lethality was dependent on water quality conditions such as total hardness and dissolved oxygen. In another acute toxicity study (Brown and Dalton 1970), a 48-hr LC₅₀ of 0.75 mg/l copper was reported for 1-year old rainbow trout (in hard water under semistatic conditions). Lloyd (1961) found a 72-hr LC₅₀ of 1.1 mg/l for rainbow trout, also with hard water (320 mg/l as CaCO₃). With soft water (15-20 mg/l as CaCO₃) the 72-hour LC₅₀ for rainbow trout was 0.44 mg/l. In another study, investigators found a 96-hr LC₅₀ in hard water (290-310 mg/l as CaCO₃) of approximately 0.9 mg/l for rainbow trout and Chinook salmon (Calamari and Marchietti 1973). Differences in results among the above experiments appear primarily related to water quality variables, especially total hardness.

Calamari and Marchetti (1973) who reported a 14-day LC₅₀ value of 0.87 mg/l copper, slightly lower than their 96-hr LC₅₀ have investigated Chronic/Sublethal effects of chronic exposure to copper in rainbow trout. Chapman (1977) reported a 200-hr LC₁₀ (lethal concentration for 10 percent of the population) to range from 0.007 to 0.019 mg/l copper for rainbow trout. In waters of intermediate hardness (100 mg/l as CaCO₃), Goettle et al. (1971) found the maximum acceptable concentration (reflecting little or no mortality) to rainbow trout in chronic bioassays to be between 0.012 and 0.019 mg/l copper.

Spawning, growth, and long-term survival of freshwater fish species are apparently affected at total copper concentrations between 5 and 40 ug/l in waters low in organic complexing matter. Lett et al. (1976) studied the effects of copper on appetite, growth, and proximate body composition of the rainbow trout. The initial effect of copper was a cessation of feeding, with a gradual return to control levels, the higher the copper concentration, the slower the return of appetite. Growth rates were depressed by copper but recovered with appetite to approach those of control fish after 40 days. Assimilation efficiency was unchanged, indicating that depressed growth represented a response to appetite suppression rather than a decreased ability to digest.

McKim and Benoit (1974) exposed brook trout to sublethal concentrations of copper from yearling through spawning to 3-month old juveniles over a 1.5 year period to determine a "no effect" concentration. No adverse effect on survival, growth, or reproductive capacity was detected in the second generation of fish from the parental stock that had previously been exposed to concentrations of 0.0094, 0.0061 and 0.0045 mg/l copper.

Salmonids have been observed in both laboratory and field situations to avoid copper (Chu et al. 1978); a threshold concentration of 0.0023 mg/l copper was estimated for Atlantic salmon. Furthermore, the olfactory response of rainbow trout to copper sulfate was shown to be depressed (Hara et al. 1976). Concentrations of less than half of the 96-hr lethal threshold value (about 0.024 mg/l) caused a marked increase in the number of spawning adult coho salmon migrating downstream without spawning. Lorz and McPherson (1976) also found reduction in the downstream migration of juvenile coho salmon after a long-term exposure of 0.005 mg/l copper, or short-term exposure to 0.030 mg/l copper.

Both marine and freshwater fishes respond to copper with periodic involuntary spasms which are similar to those of Wilson's disease (symptoms: spasmodic muscle contractions and quivering in mammals)(Benoit 1975, Baker 1969). An excess of unbound copper in the blood stream characterizes Wilson's disease. Copper was not shown to have an adverse effect on the immune response of rainbow trout.

Adult bluegills accumulated copper in the liver at concentrations lethal to larvae, the most sensitive life stage of this species. In brown bullheads (*Ictalurus nebulosa*), gill and liver tissue concentrated copper when fish were exposed to 0.027 to 0.035 mg/l for 20 months. (See Appendix G, pp.7-11 for additional technical information.)

Additional Information The synergistic effects of copper and chemical pollutants on fish have been largely ignored with the exception of the effect of mixtures of copper and zinc. Most investigations have been restricted to laboratory studies where the effects of each metal can be evaluated. Lloyd (1961) investigated the toxicity of mixtures of copper and zinc sulfate in hard and soft water on the survival time of rainbow trout. At low concentrations, toxic effects of the mixture were additive, but at higher concentrations a synergistic effect was observed.

Bioconcentration factors (BCF) for copper (only) range from 88 for the hard-shelled clam (*Mercenaria mercenaria*) to 2,000 for the green alga (*Chlorella vulgaris*) (Westerdahl and Getsinger 1988). A BCF of 290 was measured for the fathead minnow (*Pimephales promelas*) (USEPA 1980). Winner (1985) observed BCF values for the zooplankton *Daphnia magna* ranging from 1,200 to 7,100 (a value of 100 is usually regarded as a significant factor). Thus, there is a high probability that copper will bioaccumulate in aquatic organisms. Increased body burdens of metals would be of special interest to those involved with harvesting of aquatic resources for human consumption (Chu et al. 1978).

The significance of biological accumulation is probably greatest if copper is further concentrated by successive trophic levels of organisms (biomagnification). For example, the movement of copper from plant through primary herbivore, carnivore, and detrital feeders may result in further concentration. However, measurements of copper accumulation suggest that biological magnification of copper through the food chain does not occur (Krummholz and Foster 1957, Mathis and Cummings 1973, Leland and Kuwabara 1985). They noted decreasing copper concentrations among higher trophic levels and state that the classic idea of food chain enrichment, where the highest trophic levels contain the highest toxicant concentrations, does not hold for most heavy metals.

Threatened and Endangered Species Treatment with copper compounds is not expected to affect submersed or emersed plant species federally listed as rare, threatened or endangered. Given the known toxicity of copper compounds to aquatic life, primarily fish, and given the recent ESA listings of several salmonid species in Washington State waters, Ecology's Water Quality Program made a policy decision to disallow the use of copper in salmon-bearing waters in May 1999. This decision affects all waters of the state utilized by salmonids and is currently going through the administrative process to become a formal, written policy. Applications for short-term modifications to water quality standards are reviewed on a site-

specific basis for rare, threatened, or endangered species listed by US Fish and Wildlife, and "proposed sensitive" plants and animals listed by Washington State National Heritage Data System.

Water, Land and Shoreline Use

Aesthetics Use of copper would result in decreased phytoplankton concentrations, increased water clarity, and decreased populations of some species of zooplankton. However, fewer numbers of zooplankton and increased phosphorous recycling may result in subsequent rebound blooms of algae. Generally, decreased abundance of undesired algae would positively affect visual and olfactory aesthetics of the treated water body. (See Habitat section).

Recreation There are no swimming restrictions associated with use of copper compounds. Copper treatment can temporarily increase water clarity, which would improve conditions for swimming in some lakes. However, recreational areas may be closed for a few hours during treatment. Some fish may be affected at treatment concentrations.

Agriculture Copper has been known to be essential for certain fungi since 1927 and for the normal growth and development of green plants since 1931. The requirements of plants for copper are very low; however there are many instances of naturally occurring copper deficiency. Copper toxicosis in plants is almost never observed under natural conditions, but has occurred on mine spoils or where fungicides have been used excessively.

Agricultural chemicals such as pesticides and chemical fertilizers are widely used for efficient crop production and are potential sources of copper in runoff or in sediments (Chu et al. 1978). Copper sulfate is widely used in orchards, and to control helminthiasis (worms) and infectious podermatitis in cattle and sheep. Copper compounds are also used as fungicides, molluscicides, and in some insecticides.

Copper is generally added to the soil as a micronutrient at 2-50 lbs./acre for fruit trees, onions, leafy vegetables, forage grasses, corn, sorghum, and small grains. Dosages as high as 3 kg copper/ha (copper sulfate, copper EDTA, copper lignin sulfonate, or copper flavonoids) have been sprayed on soils to correct for copper deficiency.

4. Mitigation

Potential significant adverse environmental impacts associated with the use of copper to control algae may include increased nutrients available for additional algae growth, accumulation of copper in sediments, reduced dissolved oxygen levels, and chronic and acute impacts on aquatic organisms (fish and invertebrates). The potential for impacts is dependent upon water chemistry, treatment concentration, and the number of applications to a water body over time; mitigation measures should be used to reduce or avoid these impacts.

Copper herbicides are available in two different types of formulations: copper sulfate compounds and chelated or complexed copper compounds. The EPA label for Kocide (a copper sulfate formulation) recommends a copper concentration for treating algae ranging from 1/4 ppm (.25 ppm) to 2 ppm copper. A Cutrine-Plus (chelated copper) fact sheet states that Cutrine-Plus controls algae at 0.2 to 0.4 ppm copper.

Both copper sulfate and chelated copper compounds have been shown to be acutely and chronically toxic to invertebrates and vertebrates at the recommended application rates specified above. Additionally, copper only temporarily reduces algae populations and may alter algae composition from green to blue-green.

Also, nutrients from decaying algae become available for new algae growth and repeat copper sulfate treatments have been shown to accelerate phosphorous recycling from a lake bed.

Both this technical review and the EPA registration label reveal that copper sulfate at the treatment concentration may cause significant reduction in populations of aquatic invertebrates and plants. The EPA label also states that trout and other fish species may be killed at recommended application rates. Copper is more toxic both in soft water, as determined by the content of calcium carbonate in water, and in acid (low pH) waters. According to EPA (1985), at a water hardness of 290 ppm, the LC₅₀ for rainbow trout was 3.6 ppm; an LC₅₀ of 0.032 ppm was noted when hardness was maintained at 40-48 ppm.

Though copper toxicity generally decreases as water hardness increases, Ecology does not have adequate information to determine the level of water hardness that would totally buffer adverse effects of copper. Generally, water in lakes in Eastern Washington is harder, providing a greater buffering effect than the softer waters of Western Washington.

Temperature has also been shown effect copper toxicity. EPA reports that "at 7 degrees centigrade copper sulfate was moderately toxic (LC₅₀ = 1.5 ppm) to rainbow trout, while at 12 degrees centigrade, copper sulfate became highly toxic (LC₅₀=0.2 ppm)."

Registration labels for chelated (complexed) copper bear warnings similar to those for copper sulfate. They also provide a hardness threshold of 50 parts per million (mg/l) of calcium carbonate (i.e. the labels state that copper shall not be applied to water with a hardness less than 50 mg/l). Even with this restriction the label states that the product may be toxic to fish at treatment concentrations.

The following mitigation measures should provide some level of protection to aquatic systems.

1. As noted in the "impacts" section of this EIS, copper at very low concentrations has been shown to affect trout and salmon during various life stages. Even in waters of intermediate hardness (100 mg/l as CaCO₃) the maximum acceptable concentration reflecting little or no mortality to rainbow trout in chronic bioassays ranged from 0.012 to 0.019 mg/l (ppm) of copper.

In general, information indicates that it is not advisable to use copper in waters where salmon or trout are present in any life stage (including eggs, fry, smolt, or adults). Permits may be denied if impacts to fisheries cannot be avoided. Permits may also restrict application of copper compounds to a period of time when fish are not present in the waterbody proposed for treatment.

Permits may also be conditioned to limit the size and/or location of the treatment area. Special precautions must be taken if it is determined that treatment is necessary in waters where sensitive species are present. The area of application should be limited so that the overall concentration in the water body (assuming total mixing) would be less than 0.012 ppm (the lowest concentration at which we know that impacts to fisheries occur). For example, at a treatment concentration of 1.0 ppm copper, less than 10% of the total volume of the water body should be treated (this would reduce the whole-lake concentration to a level below 0.012 ppm).

In some deep lakes, treatment could be staged to provide 100% coverage of surface waters. Staging would allow treatment of one/half (or some percentage less than one half) the surface area so that sensitive species could escape to the untreated portion of the lake. After waiting an appropriate length of time, other portions of the lake could be treated.

2. Water hardness measured in milligrams per liter as calcium carbonate (CaCO₃), must be submitted with the permit application. Per the EPA registration label, use of copper compounds will not be permitted in water with a calcium carbonate hardness less than 50 mg/l. The potential for impacts to occur at a hardness greater than 50 mg/l may be evaluated during the permit review process, and a permit may be conditioned or denied based on this evaluation.

3. The pH of water proposed for treatment must be submitted with the permit application. Copper complexes should not be used "where pH of water or spray environment is below 6, because of copper ion formation and subsequent toxicity to fish". Copper will not be permitted for use in waters with a pH of 6 or less if waters are fish bearing or are considered environmentally significant. Of the 25 lakes surveyed through Ecology's Volunteer Lakes Program, several had a pH below 6 at some point in the year (Ecology, 1990).

The permit may also limit the allowable change in pH resulting from use of copper herbicides, and may stipulate that the pH be measured before and after treatment.

4. As noted previously, copper has been shown to be more toxic at higher water temperatures than at lower temperatures. For this reason, the permit applicant may be required to submit information about waterbody temperature and this information may be factored into the permit decisions. Use of copper products may be restricted if water temperature exceeds a certain threshold, recognizing that temperature within a waterbody may be highly variable depending on depth and season.

5. Unless removed from a system, copper may precipitate and become incorporated into the sediment regardless of the formulation used (copper sulfate or chelated copper). Upon receipt of a request to apply copper-based herbicides, Ecology will evaluate the proposal for potential sediment impacts. Based on this review, Ecology may require that sediment in the water body proposed for treatment be tested to determine the concentration of copper in sediment. Ecology will review results of the sediment analysis to determine if addition of copper herbicides to the system would be inconsistent with Ecology's sediment anti-degradation policy.

A permit may be denied if Ecology determines that the use of copper would be inconsistent with this policy or other provisions of Chapter 173-204 WAC, or if existing copper concentrations in sediment are determined to be biologically significant. Chemical and/or biological testing before or after copper herbicides are used may also be required to establish impacts associated with this discharge.

In evaluating copper sediment levels, in lieu of adopted criteria, Ecology will consider existing criteria, studies, and ongoing research. For example, the marine sediment criteria for copper is 390 mg/kg dry weight [parts per million (ppm) dry]. Agencies in Canada and the U. S. have established freshwater-sediment copper criteria that were derived through various mechanisms and range from 16 ppm to 110 ppm.

6. In consideration of copper toxicity in aquatic environments and persistence in sediment, Ecology may elect to limit the number of times copper may be used per season and over time, e. g. only once per season and no more than three consecutive seasons. Segmented treatment that resulted in one full coverage of a waterbody would be considered "one treatment".

7. To reduce the potential for impacts to the aquatic environment, Ecology may limit treatments to lakes with an algae problem that exceeds a "severity" threshold. The severity of an algae problem can be determined, in part, by the depth of light penetration (water clarity) as measured by secchi disc readings, measurement of epilimnetic chlorophyll a, and phytoplankton abundance and composition.

8. Ecology, in cooperation with applicators and other interested parties, will evaluate whether chelated copper compounds can achieve results desired by the applicator at a lesser concentration than copper sulfate. Depending on the results of this evaluation, the permitter may choose to encourage the use of chelated copper compounds instead of copper sulfate. Further research may result in additional restrictions on the use of copper sulfate.

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I. Diquat

1. Registration Status

The 1992 SEIS stated that Diquat would not be permitted for use in Washington waters until critical information is available. We considered permitting Diquat with appropriate mitigation, which would have included extensive and expensive residue sampling. Given that there is a less toxic contact herbicide available (endothall) and that we have limited resources available to establish monitoring requirements or review monitoring data, Ecology determined that requiring such mitigation would not be feasible. Diquat was reviewed in the 1980 Draft and Final Environmental Impact Statements on Aquatic Plant Management in Washington State and additional information and analyses of diquat impacts are available in these two documents. Diquat will be assessed again in 2001.

2. Description

Diquat dibromide [6,7-dihydrodipyrido(1,2-a:2',1'-c) pyrazinediium dibromide] is a dipyridylum compound related to quaternary ammonium compounds (Crafts 1975 in Westerdahl and Getsinger 1988). All diquat formulations are liquid bromine salts. Diquat is a dark brown, odorless liquid of molecular weight 344, and is water soluble (568 mg/l or higher) (Hunter et al. 1984). It was first sold in the U.S. in 1967.

Diquat is a contact herbicide that kills both submerged and emerged plants. Watermilfoil is among the plants for which the Diquat label lists an application rate. Because the action of Diquat is dependent on sunlight, control of plants above water occurs more quickly (within 10 days), than does control of plants below water (30 to 40 days). Diquat is a non-selective, broad spectrum contact herbicide with only local translocation. It is absorbed through the cuticle of the leaf. Diquat acts by interfering with photosynthesis, creating rapid inactivation of cells and cellular functions through the release of strong oxidants. Phytotoxic effects of diquat on above-surface foliage can be seen within one hour of treatment in bright sunlight.

Diquat is applied with surface spray in early season and with subsurface injection when submerged weeds have reached the water surface. In firm sandy-bottom lakes with slow moving water, diquat is placed one to two inches above the lake bottom with weighted trailing hoses. Diquat is also subject to photochemical degradation. Sorption and microbial degradation are the major fate processes affecting diquat persistence (Simsiman et al. 1976). Diquat has no vapor drift (although spray drift to crops that may be damaged should be avoided).

Data Gaps *Chemical Watch in Pesticides and You* (1986) states that EPA's data base on Diquat is replete with numerous data gaps in the area of exposure and environmental fate. Furthermore, a "no-effect" level has never been determined. Summary of Data Gaps:

- Toxicological
 1. Acute oral toxicity (rats)
 2. Acute dermal toxicity (rabbits and rats)
 3. Acute inhalation toxicity (rats)
 4. Primary eye irritation (rabbits)
 5. Primary dermal irritation (rabbits)
 6. 21-day dermal toxicity (rabbits and rats)
 7. 21-day inhalation toxicity (rats)
 8. Chronic toxicity (dog)
 9. Mutagenicity studies

10. Additional data for rat chronic feeding/oncogenicity and mouse oncogenicity (tumor causing)
- Ecological Effects
 1. Avian dietary - LC₅₀
 2. Freshwater fish - LC₅₀
 3. Aquatic LC₅₀ (invertebrates)
 4. Aquatic LC₅₀
 5. Phytotoxicity
 - Environmental Fate/Exposure
 1. Degradation studies except hydrolysis
 2. Metabolism Studies
 3. Mobility Studies
 4. Dissipation studies
 5. Accumulation Studies
 6. Re-entry studies
 - Product Chemistry/Residue Chemistry
 1. Product ID and composition
 2. Analysis and certification of project ingredients
 3. Product chemistry
 4. Selected residue studies

In the concentrated form, Diquat may be harmful or fatal if swallowed, inhaled, or absorbed through the skin. The probable oral lethal dose of diquat to humans is between 50-500 mg/kg, or between 1 teaspoon and 1 ounce for a 154 pound person (Gosselin et al. 1984). The acute oral LD₅₀ (rat) was 600 (female) and 810 (male) mg of formulation/kg of body weight. Concentrated diquat can cause substantial but temporary eye injury and skin irritation. Diquat contains ethylene dibromide (EDB) as an impurity in very small quantities. EPA, Office of Pesticide Programs, has determined that because of low levels present, EDB is not expected to present any hazard if the product is handled according to label precautions.

3. Environmental and Human Health Impacts

Earth

Soils and Sediment Diquat tightly adsorbs to clay. A reaction between the double positively charged diquat cation and clay minerals present in sediments forms complexes with negatively charged sites on the clay minerals (Westerdahl and Getsinger 1988). Diquat may even insert into layer planes of expandable clay minerals such as montmorillonite. Diquat also binds to soils and sediments by incorporation into humus and by normal Langmuir-type (physical) adsorption onto organic matter and particles.

Diquat persists indefinitely but has been shown to bind rapidly and tightly to some soil particles. The binding capacity of Diquat may be variable depending on available particle sites, soil type, and other factors. Binding of diquat to sandy sediments might be as much as 10 times slower than to clayey, silty, or loamy sediments (Ecology and Environment, Inc. 1991). In muck soils, it may take several days for diquat initially adsorbed onto relatively weak adsorption sites on organic matter to be transferred to the strong adsorption sites on clay minerals (Valent U.S.A. Corporation, 1989).

Diquat is not considered bioavailable when bound (Simsiman et al. 1976). Diquat is so firmly adsorbed to clay minerals that it can only be displaced by extremely rigorous treatments, such as boiling the soil with

12N - 18N sulfuric acid for several hours. This process destroys the clay structure and organic matter, thereby eliminating adsorption sites.

Diquat spray that lands on leaf surfaces undergoes extensive photochemical degradation. When the desiccated plant is later incorporated into soil, degradation of the photoproduct residue occurs through microbial degradation ultimately to carbon dioxide.

There is no major degradation of diquat itself after direct application to soil. In large pot tests using several soil types, there was no significant degradation of diquat over a 2.5-year period. In the field, studies have shown no significant decrease in diquat residues in various soil types over 4.5 years. Photochemical degradation products of diquat formed on grass are not accumulated in soil when the sprayed sward is later incorporated into the soil (Valent U.S.A. Corporation, 1989).

In the laboratory, approximately 80-90% of diquat added to a water-sediment flask system was sorbed to the sediment within 2 days (Simsiman and Chesters 1976). In the field, no evidence of desorption of diquat from pond sediments 8.5-100 weeks after treatment with 2.5 ug/ml diquat was observed by Grzenda et al. (1966). However, diquat persisted in sediments longer than 160 days after treatment in another pond study (Frank and Comes 1967).

Air

Air quality Adverse impacts to air quality are expected to be minor, such as a small amount of exhaust emission associated with the use of application equipment. The vapor pressure of diquat dibromide is too low to be measured, thus there is no possibility of a vapor hazard (Valent U.S.A. Corporation 1989). Furthermore, little aerial drift or overspray is expected if label warnings are followed, and no aerial drift is expected when herbicide application is performed with subsurface applicator devices.

Release of Toxic Materials Diquat does contain ethylene dibromide (EDB) in very small quantities as an impurity. If the concentrate is spilled during formulating operations and allowed to stand, it can dry to a highly irritating dust. Symptoms of inhalation overexposure to spray mist or dust may include headache, nosebleed, sore throat, and coughing. Therefore, spills of diquat should be cleaned up immediately. The spill should be covered with a generous amount of absorbent (clay or loam soil), and the absorbent mixed with a broom then swept. Finally, the spill area should be scrubbed with detergent and water.

Water

Surface Water Use of diquat in the treatment of dense weed areas can result in oxygen loss from decomposition of dead weeds. A review of diquat (Dynamic Corporation, 1985) contains a summary of various diquat dissipation rates and concludes that "Residues were highly variable within and between tests." In one case, "tolerance-exceeding [tolerance is 0.01 ppm] residues occurred in a single sample collected 22 days after treatment with 5 lb cation/A (to give 1 ppm). This particular test (T-697) also gave very high residues on days 1, 5, and 8 compared to similar tests and may reflect the influence of limnological factors unique to the test pond, or environmental and meteorological factors peculiar to this test."

In another test, residues in ponds receiving one treatment at 1 ppm ranged from 0.13 - 0.16 ppm on day 10, and were non-detectable (<0.01 ppm) from day 21 to day 168. However, "two ponds both treated once at 3 ppm were 0.07 and 0.12 ppm 21 days post-treatment, and in one pond, 0.01 ppm 42 days post-treatment."

In conclusion, the authors state: The available data demonstrate that the nature and magnitude of residues of diquat in natural waters are highly variable and unpredictable. Insufficient information is available to permit us to assess the importance of the numerous limnological and other environmental factors that may influence the rate of dissipation of residues following treatment. We therefore recommend that use be restricted to the U.S. Army Corps of Engineers or other federal or state agencies competent to ensure that all restrictions on the use of diquat-containing water are enforced and that approved analysis of water samples is properly carried out. Further, we recommend that the 21 CFR 193.160 be amended to read "...for 14 days post-treatment and until approved analysis shows that the water does not contain more than 0.01 ppm of diquat..."

Ground Water Diquat strongly binds to some soils totally and irreversibly. When diquat passes into the tightly bound and biologically unavailable condition, it does not persist in the environment for long periods in the unbound and biologically available state. Adsorption is complete and irrespective of pH. When applied terrestrially under field conditions, it was observed that the greatest concentration of diquat is in the top inch of soil, even after 4.5 years. In this situation, movement could only occur when soil particles themselves are eroded by "runoff" water. Where erosion does take place, diquat is not released into the terrestrial or aquatic environment.

However, Extoxnet (1987) states that there is evidence that diquat has the ability to saturate all available adsorption sites on soil clay particles. Ground water would be affected if soil adsorption sites become totally saturated. Extoxnet suggests that more research is needed for a better understanding of diquat effects on ground water.

Public Water Supplies Diquat is effectively adsorbed and removed from water by the activated carbon or clay minerals used in water treatment plants (0.05 ug/g diquat could be reduced to 0.005 ug/g with 7 mg carbon or 1 mg bentonite per liter of water) (Parkash 1974, Zarins 1965). Diquat could impact drinking water drawn directly from a treated water body.

Diquat is moderately toxic, with large doses (acute exposure) causing accumulation of water in the gastrointestinal tract with corresponding dehydration of blood and other tissues and organs. Long-term lower-level exposure (chronic exposure) caused corneal opacity and cataracts in animals. Diquat is teratogenic (e. g. causes birth defects), but to date these effects have only been detected when diquat was administered interperitonally or by injection. Data regarding mutagenicity conflicts, with both positive and negative findings using the same bioassay system. Mutagenicity is the property of a substance to induce changes in the genetic complement in subsequent generations. The EPA assessment of carcinogenicity is pending until additional tests are available, though a number of studies indicate that diquat is not carcinogenic. Diquat has been reported to alter male mouse fertility (Doull et al. 1980).

According to a recent review of diquat toxicity criteria, the only existing EPA toxicity criterion is an oral reference dose for assessing chronic (long-term) exposures to diquat, which has been established at 2.2×10^{-3} (Schoof, R. A. Personal communication. February 26, 1991). EPA has proposed a drinking water equivalent level for diquat of 0.077 mg/L (or ppm) and a maximum contaminant level goal of 0.02 mg/L (or ppm).

While available data indicate that a single application of diquat to intact human skin results in little absorption (0.3%), there is concern about penetration of broken or abraded skin. (EPA, 1986).

EPA characterizes Diquat-dibromide data as not adequate to fully assess acute toxicity and insufficient to assess environmental fate. EPA has established a 0.01 ppm tolerance for residues of diquat dibromide in potable water and has designated diquat as a restricted use pesticide.

Plants

Non-target plants Diquat is a contact type, nonselective herbicide absorbed by foliage, and therefore the hazard to non-target plants is great. Diquat causes rapid inactivation of cells and cellular functions through release of strong oxidants (Westerdahl and Getsinger 1988). Diquat controls many submersed aquatic macrophytes and some types of filamentous algae in static and low-turbidity water (Klingman et al. 1975). Plants are sensitive to diquat in soil solution at concentrations as low as 0.01 ug/ml.

Ecology does not support removal of non-noxious emergent (wetland) species except in controlled situations where the intent is to improve low quality (Category IV) wetlands or situations where wetlands have been created for other specific uses such as stormwater retention.

According to EPA, the metabolism of diquat in plants has not been adequately described and other metabolites of concern may be discovered (1986). Information available indicates that diquat undergoes rapid photochemical degradation on plant surfaces and in water exposed to sunlight. 1,2,3 4-tetrahydro-1-oxopyrido-[1,2-a]-5-pyrazinium ion (II, TOPPS) is the major degradation product. On further irradiation, this compound is degraded to picolinamide (III) and then via picolinic acid (IV) to volatile fragments (Smith and Grove 1969). A second, minor degradation pathway results in the formation of the diones (V) and (VI). The monopyridone (VII) is formed only to a very limited extent.

Information from the manufacturing company (Valent U.S.A. Corporation) states that diquat can control the following plant species.

Submersed weeds	Bladderwort (<i>Utricularia</i> spp.) Naid (<i>Najas</i> spp.) Coontail (<i>Ceratophyllum demersum</i>) Watermilfoil (<i>Myriophyllum</i> spp.) Pondweed (<i>Potamogeton</i> spp. except <i>P. robbinsii</i>) Elodea (<i>Elodea</i> spp.) Hydrilla (<i>Hydrilla verticillata</i>)
Floating weeds	Waterhyacinth (<i>Eichhornia crassipes</i>) Salvinia (<i>Salvinia rotundifolia</i>) Waterlettuce (<i>Pistia stratiotes</i>) Pennywort (<i>Hydrocotyle umbellata</i>) Duckweed (<i>Lemna</i> spp.)
Marginal weeds	Cattail (<i>Typha</i> spp.)
Algae	<i>Pithophora</i> spp. <i>Spirogyra</i> spp.

Animals

Freshwater Invertebrates MacKenzie (1971) reviewed information on the effects of diquat on aquatic invertebrates and concluded that diquat does not affect these organisms at rates used for vegetation control. Diquat did show a transitory effect on the zooplankton *Daphnia* (Gilderhus 1967), and the median immobilization concentration to *D. magna* was 7.1 ppm (Crosby and Tucker 1966). Studies on estuarine organisms in Florida have shown no adverse effects on oysters, shrimp or fish (Wilson and Bond 1969).

Diquat had no direct effect on aquatic insects and related animals in a pond study (Hilsenhoff 1966). However, decreased pond weed after treatment did lead to migration of some species to shoreline vegetation. After loss of aquatic vegetation with 0.5 ppm diquat, the decaying vegetation appeared to benefit certain benthic organisms such as *Oligochaeta*, indicated by increased numbers (Tatum and Blackburn 1962). However, this concentration acted as either direct or chronic poison to chironomids.

Dragonflies, damselflies, and tendipedids survived diquat concentrations 40 times the maximum field application rate, but the amphipod *Hyaella* was highly sensitive to diquat (Wilson and Bond 1969). Diquat also showed an acute toxicity to cladocera, although cladocera populations returned to normal levels after diquat disappeared from the water (Gilderhus 1967).

Vertebrates Information on effects of diquat dibromide on birds indicates that it ranges from nontoxic to moderately toxic, depending on the bird type tested (EPA 1986). Diquat's acute oral LD₅₀ in twelve young male mallard ducks was 564 mg/kg (Hudson 1984). The LC₅₀ for mallards was >5,000 ppm and for pheasants was 3,600 - 3,900 ppm (Pimental 1971). In a study found by EPA to be scientifically sound but not meeting EPA's guidelines for an avian reproduction study for the registration standard, reproductive testing of bob white quail revealed that "at the 5 ppm Cation [Diquat] treatment level, a statistically significant difference (p. < .01) was observed in the body weights of both the hatchlings and the 14-day old survivors. The study author concluded "while statistically significant, the actual effect was very slight, and it is not considered to be biologically meaningful."

Fish Studies of diquat residues in fish exposed to solutions of diquat have shown 1) that there is no accumulation above the external concentration of diquat, and 2) that as residues in the water decrease, so do residues in fish (Valent U.S.A. Corporation 1989). When trout, carp, or goldfish were maintained for up to seven days in water containing diquat at 1 ug/ml, residues in fish were smaller than in the surrounding water and were found mainly in non-edible portions (skin and viscera).

Under field conditions, diquat residues in fish, oysters, and clams did not exceed the applied treatment level. Residues in fish decreased with time, normally over several weeks. The maximum residue observed in the edible portion of fish, oysters, and clams was 0.02 ppm (Cope 1966).

The decrease in residues in fish lags behind those in the water. When trout were immersed in solutions of 0.5 ppm and 1.0 ppm diquat for 16 days, the highest levels reached in the whole fish were 0.4 and 0.6 ppm, respectively. These levels slowly returned to non-detectable after returning the fish to fresh water. Similar results were obtained with goldfish.

In 13 experiments, diquat did not cause direct mortality to any fish species at 1.0 ppm or below (MacKenzie 1971). The greatest concentration of diquat allowed by the label would equate to an initial in-water diquat concentration of 1.5 ppm.

Diquat is used to treat disease in fish at hatcheries, and for the species tested did not affect the breeding rate of fish or cause mortality in juveniles (Gilderhus 1967). Rates of 1 ppm diquat applied up to 3 times and 3 ppm applied once or twice, with 8-week intervals between applications, had no adverse effect on hatching and growth rates of bluegills in seven different pools. Channel catfish fry were not affected at 10 ppm diquat and bluegill fry were not affected at 4 ppm diquat. Largemouth black bass fry were more sensitive and were affected at levels greater than 1.0 ppm at 22.5°C and at 0.5 ppm at 26.0°C (Jones 1965).

Decaying vegetation caused by diquat treatment will deplete oxygen in the water. In some circumstances, such decreased oxygen will affect fish survival. Therefore, only 1/3 to 1/2 of dense weed areas should be treated at a time, with a 14 day waiting period between treatments.

Mammals Cows are particularly sensitive to diquat dibromide (Hartley and Kidd 1983). Additionally, diquat at levels 1 to 2 ppm was highly embryotoxic to the clawed frog (*Xenopus laevis*). The bioconcentration factor (BCF) for diquat is low: <1-62 (Westerdahl and Getsinger 1988). When diquat is administered over a prolonged period, there is no buildup in animal tissues of the herbicide. (Valent U.S.A. Corporation 1989).

Threatened and Endangered Species Treatment with herbicides has the potential to affect submersed and emersed plant species federally listed as rare, threatened, or endangered. These species may be aquatic or may occur along the banks of waterways. The spotted frog, a state and federal candidate for listing as endangered, threatened, or sensitive, may also be affected.

Water, Land and Shoreline Use

Aesthetics Removal of aquatic vegetation may be viewed as either positively or adversely impact on aesthetics, depending on the attitude of the observer.

Recreation Swimming is restricted for 24 hours after treatment with diquat. However, removal of dense vegetation from areas used for swimming would most likely improve swimming conditions water skiing and boating. Navigation to and from fishing areas would be improved after removal of dense aquatic vegetation. Habitat for recreational fish species such as largemouth bass would be reduced with treatment, although habitat for recreational fish species such as trout may be improved with treatment.

Parks and Recreation Though the EPA label for Diquat has no fishing restrictions, a recent risk assessment recommended waiting, under worst case conditions, 6 days after treatment before taking fish for consumption. This 6-day waiting period was further modified by statements indicating that diquat may persist as much as 10 times longer than data upon which the worst case waiting-period was based. The EPA registration label carries a 24-hour swimming restriction. Additionally, recreational areas may be closed for a few hours during treatment.

Removal of dense aquatic vegetation may improve recreational facilities for water sports such as swimming, water skiing, sailboarding, and boating.

Agriculture Diquat is used in agriculture for the desiccation of alfalfa, clover grain, sorghum, and soybean seed crops, and for potato plants in preharvest application in order to facilitate harvest. Diquat is also used in conservation of forage, such as prewilting for silage and preparation of standing hay.

A concern with the use of diquat is damage by drift to plants and crops. However, adequate label warnings are given which, if followed, would prevent drift from occurring.

The EPA label restricts use of diquat treated water for irrigation for 14 days after application. Phytotoxic damage to most crops irrigated with water containing 0.5 ppm diquat is unlikely; nevertheless, residues of the chemical could occur, particularly in leafy vegetables subjected to several prolonged irrigations (Davis et al. 1972 in Valent U.S.A. Corporation 1989).

Water/Stormwater Herbicides are not expected to be used to control vegetation in stormwater drainage facilities such as extended detention ponds or artificial wetlands. Some small waterbodies may overflow into stormwater drainages if water levels are increased after a rainfall (B. Miller, DowElanco Company, personal communication). Therefore water levels should be lowered slightly before treatment in ponds where there is a high potential for overflow.

4. Mitigation

Ecology determined in the 1992 SEIS that diquat would no longer be permitted for use in Washington waters until critical information is available. We considered permitting diquat with appropriate mitigation, which would have included extensive and expensive residue sampling. Given that there is a less toxic contact herbicide available (endothall) and that we have limited resources available to establish monitoring requirements or review monitoring data, we determined that requiring such mitigation would not be feasible.

Several beneficial uses of water are affected by treatment with diquat. The EPA label prohibits swimming in diquat-treated water for 24 hours after treatment; and restricts animal consumption, spraying, irrigation, and domestic use for 14 days after treatment. If activities were conducted under the supplemental diquat label, water could not be used for animal consumption, spraying, irrigation, or domestic purposes for 14 days after treatment, or until an approved assay shows that the water does not contain more than 0.01 part per million of diquat dibromide (personal communication, Michelle Leech, Agriculture, 3/1/91).

According to a recent review of diquat toxicity criteria, the only existing EPA toxicity criterion is an oral reference dose for assessing chronic (long-term) exposures to diquat, which has been established at 2.2×10^{-3} (Schoof, R. A. Personal communication. February 26, 1991). This review also states that EPA's final assessment of the carcinogenicity is pending, and studies of mutagenicity yielded both positive and negative findings. EPA has proposed a drinking water equivalent level for diquat of 0.077 mg/L (or ppm) and a maximum contaminant level goal of 0.02 mg/L (or ppm).

The initial review of diquat, as presented in the draft EIS, indicated that the bond of diquat to soil is rapid and irreversible. Based on this assumption, it was determined that the use of diquat would have minimum impacts. Subsequent to issuance of the draft EIS, Ecology and Environment (E & E) was hired to conduct a risk assessment of diquat. This assessment explains that most research to date was conducted under circumstances very favorable to rapid degradation and adsorption. E & E estimated that actual rates may be more than 10 times slower than indicated by this research.

A review of diquat (Dynamic Corporation, 1985) contains a summary of various diquat dissipation rates and concludes "Residues were highly variable within and between tests." In one case, "tolerance-exceeding [tolerance is 0.01 ppm] residues occurred in a single sample collected 22 days after treatment with 5 lb cation/A (to give 1 ppm). This particular test (T-697) also gave very high residues on days 1, 5, and 8 compared to similar tests and may reflect the influence of limnological factors unique to the test pond, or environmental and meteorological factors peculiar to this test."

In another test, residues in ponds receiving one treatment at 1 ppm ranged from 0.13 - 0.16 ppm on day 10, and were non-detectable (<0.01 ppm) from day 21 to day 168. However, "two ponds both treated once at 3 ppm were 0.07 and 0.12 ppm 21 days post-treatment, and in one pond, 0.01 ppm 42 days post-treatment."

In conclusion, the authors state that the available data demonstrate that the nature and magnitude of residues of diquat in natural waters are highly variable and unpredictable. Insufficient information is available to permit us to assess the importance of the numerous limnological and other environmental factors that may influence the rate of dissipation of residues following treatment. We therefore recommend that use be restricted to the U.S. Army Corps of Engineers or other federal or state agencies competent to ensure that all restrictions on the use of diquat-containing water are enforced and that approved analysis of water samples is properly carried out. Further, we recommend that the 21 CFR 193.160 be amended to read "...for 14 days post-treatment and until approved analysis shows that the water does not contain more than 0.01 ppm of diquat..."

In summary, the re-entry schedule developed by E & E does not incorporate potential variability and unpredictability in dissipation rates. For this reason, this re-entry schedule can not be used in isolation when determining when waters may be safe for drinking, swimming, or when fish may be safe for consumption. The variability in dissipation and absence of critical data also makes it impossible for us to assess potential effects to human health from use of diquat. We can state the Maximum Allowable Concentration (MAC) for diquat, as estimated by E & E without all necessary information, may be exceeded for long periods (possibly more than 210 days for drinking water and 60 days for fish consumption) after application of Diquat at label rates.

We are also concerned that critical health data is not available, a concern compounded by the fact that dissipation rates have been shown to be extremely variable. Additionally, diquat has been shown to be highly embryotoxic to the clawed frog which raises concerns about potential direct affects to other amphibians (including the spotted frog, which is a state and federal candidate for listing as endangered, threatened, or sensitive; and is a priority species under the Department of Wildlife priority habitats and species program). We are also concerned about potential indirect effects to the food chain. Diquat has also been shown to kill some fish species at application rates. For these reasons, Diquat will not be permitted for use in Washington waters.

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J. Endothall Formulations of Aquathol

1. Registration Status

The Washington State Departments of Agriculture and Ecology have approved Aquathol® Granular Aquatic Herbicide (EPA Reg. No. 4581-201), and Aquathol® K Aquatic Herbicide (EPA Reg. No. 4581-204), for use in control of aquatic macrophytes (plants) in lakes and ponds. Aquathol® K has also been approved for control of aquatic macrophytes in irrigation canals. Aquathol® Super K has received a Federal Registration for control of aquatic macrophytes in lakes and ponds and but as of January 2000 it is not registered for use in Washington State (Appendix D, Vol. 2, Sect. 1, p. 3).

2. Description

Endothall (7-oxabicyclo [2,2,1] heptane-2,3-dicarboxylic acid) is the active component in Aquathol® and Aquathol® K, containing 10.1% and 40.3% of the active ingredient respectively. Endothall is a contact herbicide that disrupts solute transport processes in plant cells. The mode of action of endothall is not fully understood; however, all of the hypotheses indicate that endothall disrupts biochemical processes at the cellular level. Endothall is formulated in two active ingredient forms, the dipotassium salt and the dimethylalkylamine salt. The potassium salt formulation is used in both Aquathol® and Aquathol® K (Appendix D, Vol. 2, Sect. 2, p. 3).

Data Gaps

- Soil and Sediment - Concentration of endothall in sediment due to the use of granular Aquathol® needs to be further investigated. Without well-determined values for how much endothall a given soil type removes and how rapidly, assumptions may lead to an estimated water column half-life that is too long. A knowledge of the concentration of endothall in the sediment is necessary so that an adequate Risk Quotient and evaluation can be made for sediment organisms.
- Water - It is generally believed that dissolved oxygen, ammonia, nitrite and nitrate, phosphate, iron, pH, hardness and alkalinity effect the toxicity and secondary effects of endothall but the database is far from complete. The areas of debate among scientists are whether or not the increase of nitrogen and phosphate on the death of treated aquatic weeds cause algal blooms.
- Plants - Dead and dying plants may release nitrogen and phosphorous which are rapidly taken up by unaffected plants. However, the data are not clear in this case. The planting of desirable vegetation after treatment with endothall has yet to receive serious investigation.
- Chronic Toxicity Studies for Plants and Animals - There are few well designed chronic toxicity studies that have been conducted with Aquathol®, Aquathol® K and Hydrothol®. For an ideal understanding of chronic effects, early life stage studies need to be conducted on all end use products or their technical equivalence with rainbow trout, fathead minnow and sheepshead minnow. Since Coho salmon and Chinook salmon are so important in the Northwest, Early Life Stage (ELS) studies and further smoltification studies should also be conducted with these species (Appendix D, Vol. 2, Sect. 4, pp. 66-67).

3. Environmental and Human Health Impacts

Earth

Soil Information on endothall persistence in soil can be useful in predicting its environmental fate when accidentally oversprayed on shorelines or when water levels drop in treated lakes and ponds, exposing sediment to the air. Endothall half-lives in aerobic soils with viable microbial populations ranged from less than one week to approximately 30 days. In two field tests, residues were non-detectable after 21 days. In soils suspected of not having sufficient microbial populations, or populations of microorganisms able to degrade endothall, two studies found a half-life of 166 days and persistence of residues over 0.05 ppm more than one year (Appendix D, Vol. 2, Sect. 3, p. 7).

Due to high water solubility and low soil/water distribution coefficient, Aquathol® (dipotassium endothall salt) does not adsorb well to most soils (Appendix D: Sect. 4, p. 22).

Sediment Endothall persistence in sediment has not been investigated as thoroughly as in water, but half-lives of 8 to 32 days were reported, with disappearance taking 22 to 36 days. Many reviewed studies did not address water and sediment persistence separately, but reported disappearance in the "system" as taking 1 to 26 days. Sediment persistence can be expected to be longer when granular formulations are used as opposed to liquid formulations, since granules resting on the sediment continue to release endothall over a period of days. The major product of the degradation of endothall is CO₂, the end product of microbial metabolism of endothall's carbon atoms. Small amounts of humic acid, fulvic acid, and humin have been identified. Their presence reflects the incorporation of endothall carbon into naturally occurring soil components (Appendix D, Vol. 2, Sect. 3, p. 9).

Air

Toxicity Results of rat acute inhalation toxicity studies concerning endothall technical and its various product formulations indicate that the animals displayed signs of respiratory tract irritation during the 4-hour exposures and the recovery periods. Signs of respiratory tract irritation during exposure included labored breathing, decreased respiratory rates and increased eye and nasal secretions. Immediately after exposure and during the first week of the observation period, signs of labored breathing and decreased respiratory rates persisted along with rales, eye and nasal discharge and decreased activity. No gross pathological findings were found at necropsy associated with exposure to the test material (Appendix D, Vol. 2, Sect. 5, p. 5).

Drift Inhalation exposures to endothall in aquatic herbicidal use situations basically apply to the applicator where generation of a spray mist or dust may occur (Appendix D, Vol. 2, Sect. 5, p. 6).

There have been anecdotal reports to poison control centers and the Washington State Pesticide Incident Reporting and Tracking database concerning oral, dermal, inhalation and eye exposures to the chemical. Since endothall products and spray mixes may be irritating, depending upon the concentration, reports of irritation to the eyes, skin, respiratory tract and digestive tract following overexposure would not be unexpected. Accidental swallowing or inhalation of a strong spray dilution may irritate the gastrointestinal tract to produce signs and symptoms of nausea, vomiting and diarrhea. Similarly, inhalation of a strong spray dilution may also cause upper respiratory tract irritation as evidenced by nasal discharge, coughing, sore throat and difficulty in breathing. All of these effects are expected to remit once exposure is discontinued (Appendix D, Vol. 2, Sect. 5, p. 10).

Water

Environmental Fate Endothall acid is extremely stable in water. There was no measurable breakdown in 30 days at pH 5 and pH 9. A half-life in pH 7 water was calculated to be 2,825 days. No degradation products were identified. Endothall can be absorbed or adsorbed by aquatic plants and algae, but may be released back into the water when they die. It does not undergo hydrolysis or photolysis, but rather is broken down by the action of microorganisms that utilize it as a carbon/energy source. The half-life of endothall in water is generally ranges from less than one day to about 8 days. Total persistence time in water normally varies from a day to about 35 days, although persistence to more than 62 days has been reported.

The disappearance of endothall from a lake or other natural water body is influenced by a number of environmental factors that make it difficult to precisely calculate the degree of persistence for a specific water body. Higher water and sediment temperatures will facilitate the metabolism of endothall, while cooler temperatures, such as those found at the bottom of stratified lakes, will retard it. Water pH has little effect on endothall persistence, unless it is so extremely acidic or basic as to affect the microbial community. The amount of oxygen dissolved in a water body has a direct effect on the speed of endothall metabolism since the microorganisms that break down endothall are aerobes that must have oxygen to thrive. Warmer water, aerobic decay of organic materials on/in the sediment, oxygen depletion resulting from decay of a large aquatic vegetation kill are examples of situations that can deplete dissolved oxygen. In many cases, eutrophic and even mesotrophic lakes are more likely to support large populations of microorganisms that can metabolize endothall than lakes with lower nutrient levels. On the other hand, if carbon sources are not abundant, competition for the carbon in endothall can favor the growth of the microbiota that can utilize endothall exclusively. There is disagreement among researchers as to whether adsorption of endothall to sediment increases the availability to microorganisms by concentrating it on the surfaces, or decreases the availability for metabolism due to strong binding. The variable strength of the binding, depending on the nature of the sediment, is probably responsible for conflicting findings.

Probably the most important physical process affecting endothall persistence in larger water bodies is transport of treated water away from the treated area and replacement with untreated water through lateral circulation or vertical movement of water. In this regard, the larger the lake, the more wind blowing across the lake surface, and the more water exchange through inlet and outlet streams or rivers, the more likely it is that endothall residues will be rapidly dispersed and diluted to below detection limits. In small lakes, detectable concentrations of endothall may be carried a significant distance down an outlet stream if the flow is sufficient and endothall degradation is slow. Vertical dispersion is the dominant mechanism of dilution in whole-treated lakes, while a combination of vertical and horizontal water movements contribute to dispersion and dilution in lakes treated over only a part of their surface.

Liquid formulations can be expected to result in higher initial water concentrations than granular formulations, since all of the endothall is applied directly to the water. Granular formulations generally yield higher endothall sediment concentrations and longer persistence in or on sediments due to a prolonged release of endothall from the granules. Granular formulations can therefore result in lower water concentrations that may persist somewhat longer than if liquid formulations are used (Appendix D, Vol. 2, Sect. 3, pp. 9-10).

Experiments showed that in water of pH 5, pH 7, and pH 9, endothall acid does not significantly degrade as a result of exposure to light. The acid also does not significantly degrade as a result of light exposure when applied to the surface of a soil (See Appendix D: Sect. 3, p. 5).

Water Chemistry Exposure of living plant tissue to endothall products or other herbicides usually results in secondary effects that may impact the biota. When plants start to die, there is often a drop in the dissolved oxygen content associated with the decay of the dead and dying plant material. Reduction in dissolved oxygen concentration may result in aquatic animal mortality or a shift in the dominant form or diversity of biota. There may also be changes in the levels of plant nutrients due to release of phosphate from the decaying plant tissue and anoxic hypolimnion. Also ammonia may be produced from the decay of dead and dying plant tissue and may reach levels toxic to the resident biota. Ammonia may be further oxidized to nitrite, which is also toxic to fish. The presence of these nutrients may cause an algal bloom to occur. However, if significant living plant biomass persists after treatment, the released nutrients may be removed before an algal bloom can occur. Hardness and pH will not have an impact on the toxicity of disodium endothall salts when they are used at concentrations typically found in the field (Appendix D, Vol. 2, Sect. 4, pp. 29).

Multiple Applications Sites that have never been exposed to endothall products may degrade Aquathol®, Aquathol® K and Hydrothol® more slowly than sites that have had a previous exposure history. This is because it normally takes several weeks for bacteria capable of using endothall as their sole carbon source to develop out of their lag-phase and rapidly degrade applied endothall. Rapid degradation leads to a very short half-life in non-flowing water, which is usually less than 10 days. However, if experienced degradation, sorption, and dilution factors are interacting, the field half-life in water can be less than one-day. Therefore, long-term persistence of endothall at concentrations that will cause environmental damage is not likely (Appendix D, Vol. 2, Sect. 4, p. 14).

Public Water Supply The current Federal drinking water standard is less than 0.100 mg/L for endothall products. There have been a few cases where herbicides were found in well water at concentrations that exceed Washington State's Detection limits (Appendix D, Vol. 2, Sect. 4, p. 33).

Groundwater In some situations throughout the country, dipotassium endothall has been seen in ground water where recharge areas have been treated with Aquathol® K. These recharge areas usually had porous bottoms (sand or gravel) with clay layers located below the bottom of the well shaft. Usually, water treatment plants that are located a mile or more down stream from the treatment site will not experience concentrations of endothall higher than the Federal drinking water standard due to extensive dilution and lateral mixing (Appendix D, Vol. 2, Sect. 4, p. 33).

Plants

Bioconcentration in plants is not likely for Aquathol® and Aquathol® K. Some plants appear to bioaccumulate, but tissue analysis indicates that these residues are incorporated into the leaves and stems (Appendix D, Vol. 2, Sect. 4, p. 14).

Toxicity Aquathol® K is toxic to aquatic macrophytes. Field studies using Aquathol® K at the maximum use rate eliminated milfoil and most other macrophytes for up to two growing seasons and allowed tolerant anchored macrophytic algal species to dominate the water body (Appendix D, Vol. 2, Sect. 4, p. 6).

Algae Aquathol® K, and endothal acid have a very low toxicity to algal species. At typical use rates up to 3.5 mg a.e./L (5.0 mg a.i./L) control would be expected, and in field studies anchored macrophytic algae have been shown to not be controlled and to dominate a pond for up to two years after application of this control measure (Appendix D, Vol. 2, Sect. 4, p. 6).

Animals

Bioconcentration in animals is not likely for Aquathol. Although the mosquito fish has been observed to bioaccumulate endothall at tissue elevated levels, most species of fish and aquatic invertebrate do not bioaccumulate dipotassium endothall (Appendix D, Vol. 2, Sect. 4, p. 25).

Fish Maximum field rates of Aquathol® have been shown to not adversely impact survival, growth, reproduction or nesting behavior in bluegill sunfish and largemouth bass over a two-year period. Exposure of anadromous fish to sublethal concentrations of Aquathol® K may interfere with the parr to smolt metamorphosis and result in significant mortality when smolts are subsequently exposed to seawater. After exposure to Aquathol® K in freshwater, salmon smolts may not survive a 96-hour seawater challenge. Both laboratory and field tests indicate that fish do not bioaccumulate Aquathol® by up-take from the water or by an oral exposure route. Several fish species may avoid Aquathol® K at concentrations typically encountered in the field particularly when it is mixed with dalapon. Despite this “trend” evidence for avoidance behavior, the best-run laboratory studies indicate that behavior is not significantly different between treated and controls (Appendix D, Vol. 2, Sect. 4, pp. 7, 46-47).

At the projected maximum use rate of 3.5 mg a.e./L, Aquathol® K and its surrogate test substances will not chronically impact fish. True chronic exposure will not exist in the field if treatment with Aquathol® generally does not occur more than once per year, or once every other year, in a typical water body (Appendix D, Vol. 2, Sect. 4, pp. 7-8).

Invertebrates Aquathol® K, disodium endothall salt and endothall acid have a low acute toxicity to *Daphnia magna*. At the projected maximum use rate, Aquathol® K and its surrogate test substances will not acutely impact members of this segment of the biota. However, testing of more species of free swimming biota would lend greater confidence to the risk assessment dealing with this segment of the biota. The use of maximum field rates of Aquathol® has not been shown to adversely impact the numbers or species diversity of Cladocerans (*daphnids*), Copepoda, Cyclopsida and Calanoida when these species were monitored over a growing season which lasted from May through October. Neither the direct impact of Aquathol® nor secondary effects such as decreased oxygen content or decreased surface cover by resident plants had any observable adverse impact on the free-swimming invertebrate population. The only species of aquatic invertebrate that has exhibited mortality in the field due to the indirect effect of Aquathol® K is the hydrillia fly. At concentrations of Aquathol® K that controlled Hydrilla, 74% of hydrillia flies died. However, this mortality was probably due to a reduction in habitat as the number of hydrilla leaflets decreased and not due to the direct effects of endothall (Appendix D, Vol. 2, Sect. 4, pp. 7, 46-47).

Benthic Invertebrates Aquathol® K, disodium endothall salt and endothall acid have low acute toxicity to benthic (sediment dwelling) invertebrates. At the projected maximum use rate, Aquathol® K and its surrogate test substances will not acutely impact members of this segment of the biota (Appendix D, Vol. 2, Sect. 5, p. 7).

Use of Aquathol® at the maximum projected rate will not chronically impact benthic biota. Even if the highest short-term concentration of endothall in the sediment were substituted for the chronic water expected environmental effect concentrations (EECs), the risk quotient would still be below the chronic level of concern for protection of this segment of the biota (Appendix D, Vol. 2, Sect. 5, p. 8).

Endangered and Threatened Species Sensitive, endangered or threatened species of aquatic animals needing protection through mediation include several salmon and trout species, thirteen rockfish species,

two species of dace, two species of herring, and seven species of amphibian. Other species may also be sensitive, endangered, or threatened.

Water, Land and Shoreline Use

Human Exposure Repeated daily or weekly chemical exposures for short time frames typically occur during the application of a chemical or through dietary intake of a treated food crop or water. Most human chemical exposures are either acute (one time exposure) or subchronic (exposure to a chemical for a few days or weeks). The potential for subchronic exposure to endothall would also occur when the chemical is used for aquatic weed control. Such exposures for persons in contact with recently treated water would primarily involve dermal contact with the chemical through swimming, ingesting the water or sediment, or dermal contact with treated sediments and aquatic weeds.

Inhalation exposures to endothall in aquatic herbicidal use situations basically apply where generation of a spray mist or dust may occur. However, aquatic application of endothall-containing products in compliance with label directions is not expected to result in adverse health effects following contact with treated water. Further, factors mitigating against any adverse health effects from applied endothall are the significant water dilution, poor dermal and gut absorption, rapid excretion of absorbed endothall and short half-life in water all support the conclusion that overexposure to the chemical is unlikely (Appendix D, Vol. 2, Sect. 5, p. 6).

An exposure and risk assessment of persons swimming in endothall treated water where all of the endothall swallowed is 100% absorbed and that none of the applied endothall degrades so that the aquatic concentration of the chemical remains at 5 ppm are indicative of a large safety factor. The results of the exposure and risk assessment indicate that a person could swim daily in the treated water and never reach the lowest NOEL endothall dose of 2.6 mg/kg/dy. The margin of safety (MOS) for each person is determined by dividing the calculated dose by the lowest animal toxicology study NOEL. The greater the MOS values more than 100 indicate that the chemical exposure is not expected to cause adverse health effects. It must be remembered that these calculations do not represent what happens in the real world environment. As previously discussed, degradation and dilution of endothall would occur, and the chemical eventually becomes incorporated into plant tissue or bottom sediments. Other factors that would reduce the swimmer's dose of endothall includes ~10% absorption from the stomach and rapid excretion in the urine of the absorbed chemical. Therefore, the risk calculations below represent extensive human exposure to endothall treated water are not expected to result in any adverse systemic or poisoning effects (Appendix D, Vol. 2, Sect. 5, pp. 11-12).

Swimming Lunchick (1994) conducted an exposure assessment to evaluate swimmers' exposure to endothall treated water. The exposure assessment was conducted according to EPA's standard operating procedures for swimmer exposure in treated water (EPA 1993). Lunchick calculated that the daily total dose to a person swimming in water containing 5 ppm endothall was extremely low and did not present an acute toxicity risk. The results of the assessment review revealed that 95-97% of the swimmers daily dose of endothall was due to ingestion or swallowing the treated water. The remainder of endothall exposure was 2-3% and 1% to the skin and inhalation routes, respectively. The total calculated endothall daily doses for the 6 and 10 year olds and a 70 kg adult were 5.9, 3.7 and 1.9 ug/kg/day, respectively (Appendix D, Vol. 2, Sect. 5, p. 11).

Note: Washington State Department of Health (DOH) made the following comments to Appendix D, Vol. 2, Sect. 5 of the risk assessment.

...Children playing near the shoreline may certainly be expected to play in water (and sediments) for more than 30 min./day. In the Lunchick assessment, a 6 year of child with exposure to water alone (no sediments) had a calculated dose approximately equal to one quarter of EPA's Reference Dose after only 30 minutes in the water...DOH reviewed the study of eye irritation at dilute concentrations (5ppm, 25 PPM, and 50 PPM) of Aquathol® K conducted by Gary Wnorowski at Product Safety Labs (1997). While the scoring of the results appears to be consistent with EPA criteria and the guidelines of the Consumer Product Safety Commission, DOH believes that the results warrant some degree of caution. Dilutions of Aquathol were instilled into the right eye of six rabbits per dose. The left eye served as a control. Rabbits were then returned to their cages and observed for 72 hours. In the 5 PPM exposed group, five out of the six rabbits exhibited conjunctivitis at the one-hour period. Two of the affected animals had symptoms still observable at 24 hours. All symptoms cleared spontaneously within 48 hours....Given that the treatment conditions of 5 PPM are allowed with no swimming restriction, there appears to be no margin of safety for conjunctivitis on the federal label. Although common sense would probably keep a swimmer from swimming in an area during or immediately after an application, the label is silent about any such advice to swimmers. Please consider recommending a swimming advisory for Aquathol® K and Aquathol® of 24 hours in treated areas for protection of mild eye irritation. Despite the fact that conclusions were consistent with EPA criteria, conjunctivitis at 5 PPM could be of concern to members of the public using treated water (Morrissey 2000).

Agriculture Aquathol® (dipotassium endothall salt) will not bioaccumulate in wheat, spinach and table beets. Endothall is unlikely to bioaccumulate in livestock or fish. Since the mode of application of Aquathol® is typically by subsurface injection or sinking granules, drift is likely to be minimal. When used at concentrations below 3.5 mg a.e./L (5.0 mg a.i./L) Aquathol® should not have acute effects on aquaculture, but effects on more sensitive species cannot be ruled out (Appendix D: Sect. 4, p. 38).

4. Mitigation

Use Restrictions Use according to the label. Aquatic formulations of endothall have not been evaluated for aerial applications in Washington State. When used at concentrations listed on the label (09/98) Aquathol will control the aquatic macrophytes listed on the label including milfoil (*Myriophyllum* spp.), pondweed (*Potamogeton* spp., naiad (*Najas* spp.), coontail (*Ceratophyllum* spp.), hydrilla (*Hydrilla verticillata*) and *Sparganium* spp. Aquathol® K should not be used to control species of weeds that are not specified on the label. Some species are known to be tolerant to Aquathol® including *Chara* spp., American waterweed (*Elodea canadensis*), cattails (*Typha* spp.), spatterdock (*Nuphar* spp.) and fragrant water lilies (*Nymphaea* spp.) and may become dominant after other more susceptible species have been controlled. **Aquathol® K is not an algaecide and is generally ineffective in controlling algal species.** Algal species may bloom after treatment with Aquathol® if released nutrients reach levels that can sustain algal growth.

Swimming/skiing Current labels (09/98) have dropped swimming restrictions for Aquathol® K and Aquathol® formulations that were previously listed on labels dated March 1990. All General Mitigation posting requirements apply. In addition, informational buoys should be placed around the treatment area. A 24 hour swimming advisory recommended in treated areas for protection against mild eye irritation. Swimming outside the treatment area is permitted.

Boating A 24 hour advisory is recommended for boaters entering the area of treatment for protection against mild eye irritation due to drift or dust.

Drinking/Domestic Uses Aquathol® K and Aquathol® label restrictions differ. The current label for Aquathol® K (01/98) restricts the use of waters from treated areas for watering livestock, for preparing agricultural sprays for food crops, for irrigation or for domestic purposes according to the concentration used. Please refer to the label. When used at concentrations below 3.5 mg a.e./L (5.0 mg a.i./L) Aquathol® should not have acute effects on aquaculture, but effects on more sensitive species cannot be ruled out (Appendix D: Sect. 4, p. 38).

Fisheries Exposure to wild fisheries should be avoided. Aquathol® K (dipotassium endothall salt), disodium endothall salt and endothall acid will not affect aquatic biota acutely or chronically when applied at concentrations recommended on the label. The acute and chronic risk quotients do not exceed the level of concern, Aquathol® can be used for control of aquatic weeds without significant impact to fish, free-swimming invertebrates and benthic organisms. The field data that have been collected to date confirms this observation (Appendix D, Vol. 2, Sect. 4, p. 71).

Endangered Species Levels of Aquathol® K that would be found in the environment due to typical treatment practices may interfere with the salmon smoltification process resulting in death when smolts migrate from freshwater to saltwater. Extra caution should be taken when dealing with endangered species allowing for a level of concern of <0.05 rather than the more typical value of <0.1 for non endangered species. Restrictions on season of application are warranted to protect sensitive salmon smolts from the effects of endothall products; similar restrictions may be applied to protect fish and fisheries and prevent water use restrictions during the height of the recreational and commercial fishing seasons (Appendix D, Vol. 2, Sect. 4, p. 67).

Fish Consumption Aquathol® K and Aquathol® current labels restrict use of fish from treated areas for food or feed within 3 days of treatment.

Wetlands Lakes with outflows to estuaries should be treated only at low flows or with a buffer around the outlet.

Post-treatment Monitoring If treatment occurs near domestic water sources, post treatment monitoring may be desirable.

References

Please refer to citations in Appendix D: Compliance Services International, 2000. *Supplemental Environmental Impact Statement Assessments of Aquatic Herbicides*, Volume 2: Endothall. 275 pages.

Morrissey, Barbara, 2000. Letter dated June 22, 2000, to Kathleen Emmett, Washington State Department of Ecology from Barbara Morrissey, Toxicologist, Washington State Department of Health.

K. Endothall Formulations of Hydrothol 191

1. Registration Status

Hydrothol® 191 (liquid) and Hydrothol® 191 (granular) have received Federal registration for control of algae and aquatic macrophytes in canals, lakes and ponds. They do not currently have a registration in the state of Washington for the control of aquatic algae and weeds.

2. Description

Endothall (7-oxabicyclo [2,2,1] heptane-2, 3-dicarboxylic acid) is the active component in Hydrothol® 191 used in static and flowing water to control aquatic weeds and algae. Endothall is a contact herbicide. The mode of action of endothall is not fully understood, however, all of the hypotheses indicate that endothall disrupts biochemical processes at the cellular level. Endothall is formulated in two active ingredient forms, the dipotassium salt and the dimethylalkylamine salt. These salts forms are found in five different formulated products used for control of aquatic weed species. The amine salt is formulated as Hydrothol® 191 Aquatic Algaecide and Herbicide (EPA Reg. No. 4581-174) and Hydrothol® 191 Granular Aquatic Algaecide and Herbicide (EPA Reg. No. 4581-172). Hydrothol products are used predominantly for canal treatments to control algae and submerged macrophytes (Appendix D, Vol. 2, Sect. 2, pp. 3-4).

3. Environmental and Human Health Impacts

Earth

Soil Information on endothall persistence in soil can be useful in predicting its environmental fate when accidentally oversprayed on shorelines or when water levels drop in treated lakes and ponds, exposing sediment to the air. Endothall half-lives in aerobic soils with viable microbial populations ranged from less than one week to approximately 30 days. In two field tests, residues were non detectable after 21 days. In soils suspected of not having sufficient microbial populations, or populations of microorganisms able to degrade endothall, two studies found a half-life of 166 days, and persistence of residues over 0.05 ppm of more than one year (Appendix D: Vol. 2, Sect. 3, p. 7).

Due to high water solubility and low soil/water distribution coefficient, Hydrothol® 191 [mono(dimethylalkylamine) salt of endothall does not adsorb well to most soils. Therefore the concentration of endothall in hydrosoil is rarely higher than 0.5 mg a.e./L (Appendix D: Vol. 2, Sect. 4, p. 22).

Sediment Endothall persistence in sediment has not been investigated as thoroughly as in water, but half-lives of 8 to 32 days were reported, with disappearance taking 22 to 36 days. Many reviewed studies did not address water and sediment persistence separately, but reported disappearance in the "system" as taking 1 to 26 days. Sediment persistence can be expected to be longer when granular formulations are used as opposed to liquid formulations, since granules resting on the sediment continue to release endothall over a period of days. The major product of the degradation of endothall is CO₂, the end product of microbial metabolism of endothall's carbon atoms. Small amounts of humic acid, fulvic acid, and humin have been identified. Their presence reflects the incorporation of endothall carbon into naturally-occurring soil components (Appendix D: Vol. 2, Sect. 3, p. 9).

Air

Toxicity Results of the rat acute inhalation toxicity studies concerning endothall technical and its various product formulations, indicated that the animals displayed signs of respiratory tract irritation during the 4-hour exposures and the recovery periods. Signs of respiratory tract irritation during exposure included labored breathing, decreased respiratory rates and increased eye and nasal secretions.

Immediately after exposure and during the first week of the observation period, signs of labored breathing and decreased respiratory rates persisted along with rales, eye and nasal discharge and decreased activity. No gross pathological findings were found at necropsy associated with exposure to the test material. The combined sexes LC50 for the five investigations were all within the EPA FIFRA Toxicity Category range of III for acute inhalation toxicity (Appendix D: Vol. 2, Sect. 5, p. 5).

Inhalation exposures to endothall in aquatic herbicidal use situations basically apply to the applicator where generation of a spray mist or dust may occur. However, aquatic application of endothall-containing products in compliance with label directions is not expected to result in adverse health effects following contact with treated water. Further, factors mitigating against any adverse health effects from applied endothall are the significant water dilution, poor dermal and gut absorption, rapid excretion of absorbed endothall and short half-life in water all support the conclusion that overexposure to the chemical is unlikely (Appendix D: Vol. 2, Sect. 5, p. 5).

Drift Inhalation exposures to endothall in aquatic herbicidal use situations basically apply where generation of a spray mist or dust may occur (Appendix D: Vol. 2, Sect. 5, p. 6).

There have been anecdotal reports to poison control centers and the Washington State Pesticide Incident Reporting and Tracking database concerning oral, dermal, inhalation and eye exposures to the chemical. Since endothall products and spray mixes may be irritating, depending upon the concentration, reports of irritation to the eyes, skin, respiratory tract and digestive tract following overexposure would not be unexpected. Accidental swallowing or inhalation of a strong spray dilution may irritate the gastrointestinal tract to produce signs and symptoms of nausea, vomiting and diarrhea. Similarly, inhalation of a strong spray dilution may also cause upper respiratory tract irritation as evidenced by nasal discharge, coughing, sore throat and difficulty in breathing. All of these effects are expected to remit once exposure is discontinued (Appendix D: Vol. 2, Sect. 5, p. 10).

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Environmental Fate Endothall acid does not undergo hydrolysis or proteolysis, but is broken down by microorganisms utilizing it as a carbon source. There was no measurable breakdown in 30 days at pH 5 and pH 9. A half-life in pH 7 water was calculated to be 2,825 days. No degradation products were identified. The half-life of endothall in water is generally ranges from less than one day to about 8 days. Total persistence time in water normally varies from a day or two to about 35 days, although persistence to more than 62 days has been reported. Endothall can be absorbed or adsorbed by aquatic plants and algae, but may be released back into the water when they die.

The disappearance of endothall from a lake or other natural water body is influenced by a number of environmental factors, which makes it difficult to precisely calculate the degree of persistence for a specific water body. Higher water and sediment temperatures will facilitate the metabolism of endothall, while cooler temperatures, such as those found at the bottom of stratified lakes, will retard it. Water pH has little effect on endothall persistence, unless it is so extremely acidic or basic as to affect the microbial

community. The amount of oxygen dissolved in a water body has a direct effect on the speed of endothall metabolism since the microorganisms that break down endothall are aerobes that must have oxygen to thrive. Warmer water, aerobic decay of organic materials on/in the sediment, oxygen depletion resulting from decay of a large aquatic vegetation kill are examples of situations that can deplete dissolved oxygen. In many cases, eutrophic and even mesotrophic lakes are more likely to support large populations of microorganisms that can metabolize endothall than lakes with lower nutrient levels. On the other hand, if carbon sources are not abundant, competition for the carbon in endothall can favor the growth of the microbiota that can utilize endothall exclusively. There is disagreement among researchers as to whether adsorption of endothall to sediment increases the availability to microorganisms by concentrating it on the surfaces, or decreases the availability for metabolism due to strong binding. The variable strength of the binding, depending on the nature of the sediment, is probably responsible for conflicting findings.

Probably the most important physical process affecting endothall persistence in larger water bodies is transport of treated water away from the treated area and replacement with untreated water through lateral circulation or vertical movement of water. In this regard, the larger the lake, the more wind blowing across the lake surface, and the more water exchange through inlet and outlet streams or rivers, the more likely it is that endothall residues will be rapidly dispersed and diluted to below detection limits. In small lakes, detectable concentrations of endothall may be carried a significant distance down an outlet stream if the flow is sufficient and endothall degradation is slow. Vertical dispersion is the dominant mechanism of dilution in whole-treated lakes, while a combination of vertical and horizontal water movement contribute to dispersion and dilution in lakes treated over only a part of their surface.

Liquid formulations can be expected to result in higher initial water concentrations than granular formulations, since all of the endothall is applied directly to the water. Granular formulations generally yield higher endothall sediment concentrations and longer persistence in or on sediments due to a prolonged release of endothall from the granules. Granular formulations can therefore result in lower water concentrations that may persist somewhat longer than if liquid formulations are used (Appendix D: Vol. 2, Sect. 3, pp. 9-10).

Experiments showed that in water of pH 5, pH 7, and pH 9, endothall acid does not significantly degrade as a result of exposure to light. The acid also does not significantly degrade as a result of light exposure when applied to the surface of a soil (Appendix D: Vol. 2, Sect. 3, p. 5).

Water Chemistry Exposure of living plant tissue to endothall products or other herbicides usually results in secondary effects that may impact the biota. When plants start to die, there is often a drop in the dissolved oxygen content associated with the decay of the dead and dying plant material. Reduction in dissolved oxygen concentration may result in aquatic animal mortality or a shift in the dominant form or diversity of biota. There may also be changes in the levels plant nutrients due to release of phosphate from the decaying plant tissue and anoxic hypolimnion. Also ammonia may be produced from the decay of dead and dying plant tissue and may reach levels that may be toxic to the resident biota. Ammonia may be further oxidized to nitrite, which is also toxic to fish. The presence of these nutrients may cause an alga bloom to occur. However, if significant living plant biomass persists after treatment, the released nutrients may be removed before an alga bloom can occur. Hardness and pH will not have an impact on the toxicity of disodium endothall salts when they are used at concentrations typically found in the field (Appendix D: Vol. 2, Sect. 4, pp. 29).

Multiple Applications Sites that have never been exposed to endothall products may degrade Hydrothol® more slowly than sites that have had a previous exposure history. This is because it

normally takes several weeks for bacteria capable of using endothall as their sole carbon source to develop out of their lag-phase and rapidly degrade applied Hydrothol® 191 if they have not been previously exposed. Rapid degradation leads to a very short half-life in non-flowing water, which is usually less than 10 days. However, if degradation, sorption, and dilution factors are interacting, the field half-life in water can be less than one-day. Even so, due to the extremely high toxicity of the dimethylalkylamine constituent of Hydrothol® 191, concentrations of Hydrothol® 191 although similar to those of Aquathol® may be high enough to cause observable damage to the biota (Appendix D: Vol. 2, Sect. 4, pp. 13-14, 24-25, 50).

Public Water Supply In some situations throughout the country, dipotassium endothall has been seen in ground water where recharge areas have been treated with Aquathol® K. These recharge areas usually had porous bottoms (sand or gravel) with clay layers located below the bottom of the well shaft. Usually, water treatment plants that are located a mile or more down stream from the treatment site will not experience concentrations of endothall higher than the Federal drinking water standard due to extensive dilution and lateral mixing. Endothall is not likely to be found in the water of sewage outfalls since wastewater treatment plants only process water from household waste and water runoff from street level. Due to the short half-life of endothall in water bodies, additional procedures for removing endothall from sewage outfalls or potable water systems is not necessary; however, natural bacteria have the potential to remove excessive endothall from any water system in which they are found (Appendix D: Vol. 2, Sect. 4, p. 33).

Plants

Bioconcentration in plants is not likely for Hydrothol 191. Some plants appear to bioaccumulate ¹⁴C labeled endothall at concentrations that are ~4-fold higher than environmental concentrations, but tissue analysis indicates that these residues are incorporated into natural plant constituents in the leaves and stems (Appendix D: Vol. 2, Sect. 4, p. 14).

Macrophytes Hydrothol® 191 is toxic to aquatic macrophytes. The representative species in the laboratory is *Lemna gibba* and the toxicity (EC₅₀) of Hydrothol® to *Lemna gibba* is 0.83 mg a.e./L (3.5 mg product/L). Typical use rates may be as high as 5.0 mg a.e./L (5.0 mg product/L), therefore, this macrophyte would normally be controlled under field situations. Results from controlled field studies are not available. However, the 1999 label for Hydrothol® 191 indicates that pondweeds (*Potamogeton spp.*) milfoil (*Myriophyllum spp.*), coontail (*Certophyllum spp.*), American waterweed (*Elodea canadensis spp.*), Brazilian Elodea (*Egeria densa spp.*) and horned pondweed (*Zannichelia spp.*) will be controlled with Hydrothol® concentrations in the range of 0.5 to 2.5 mg a.e./L (2.1 to 11 mg product/L). For a list of species with which efficacy has been demonstrated please see Table 2 and Appendix 1 of Section 1 (Appendix D: Vol. 2, Sect. 4, pp. 8-9).

Algae Hydrothol® has a very high toxicity to many alga species. The EC₅₀ ranges from 0.0023 mg a.e./L for the green algae (*Selenastrum capricornutum*) to >0.27 mg a.e./L for the marine diatom (*Snydra* sp.) and the green algae (*Chlorella vulgaris*). There are a number of species that are not significantly affected by concentrations higher than the typical maximum use rate of 0.2 mg a.e./L. For these species, higher use rates (up to 0.8 mg a.e./L) may be necessary for control. Experimental algae control in Lake Steilacoom was not entirely effective at concentrations up to 0.2 mg a.e./L Hydrothol® although this may be the maximum concentration that risk assessments or field evaluations indicate is safe to the biota in acute or chronic exposure (Appendix D: Sect. 4, pp. 8-9).

Animals

Bioconcentration in animals is not likely for Hydrothol® 191. Although the mosquito fish has been observed to bioaccumulate endothall at tissue levels that are ten-fold higher than environmental concentrations, most species of fish and aquatic invertebrate do not bioaccumulate dipotassium endothall (Appendix D: Vol. 2, Sect. 4, p. 25).

Fish Hydrothol® 191 has a high acute toxicity to fish. The toxicity ranges from an LC50 of 0.079 mg a.e./L for cutthroat trout to 0.82 mg a.e./L for sheepshead minnow. It is noteworthy that the cutthroat trout is a threatened species in addition to being the most sensitive species tested. Since the maximum use rate of Hydrothol® is 5.0 mg a.e./L this most sensitive species of fish within the biota will suffer adverse impact from the effects of Hydrothol® 191. For example, the risk quotient is substantially above the acute level of concern (0.1 for typical species and 0.05 for endangered species) for all species tested. $RQ = 1.4 \text{ ppm a.e.} / 0.079 \text{ ppm a.e.} = \sim 18$. Therefore, the use of Hydrothol® 191 at the maximum use rate will not be safe to sensitive species within the biota. The field use rates of Hydrothol® 191 that are considerably below the maximum rate have been shown to impact resident fish populations including channel catfish, threadfin shad, red shiner and mosquito fish adversely when used at concentrations as low as 0.20-0.5 mg a.e./L in irrigation canals for periods as short as 120-hours. Modeling indicates that these effects can be decreased to less than 10% of the resident species if concentrations of Hydrothol® 191 are kept at or below 0.2 mg a.e. for 120 hours or less and some additional mitigation measures are used. Additional mitigation could be obtained by treating canals with high suspended organic carbon and low hardness (~20 mg calcium carbonate/L). Exposure of anadromous fish to sublethal concentrations (0.2 mg a.e./L) of Hydrothol® 191 that might typically be encountered in the environment may interfere with the parr to smolt metamorphosis and result in significant mortality when smolts are subsequently exposed to seawater. The manufacturer and some Washington state applicators claim that fish are able to avoid exposure to Hydrothol® 191 and its toxic dimethylalkylamine constituent. They claim that fish may be driven away from the herbicide treatment plume if the herbicide is applied from the shore outline outward with skill and understanding of fish avoidance behavior. However, these observations are not supported by credible studies using proper controls (Appendix D: Vol. 2, Sect. 4, p.9).

The chronic toxicity of Hydrothol® 191 ranges from a chronic NOEC of 0.022 to 0.056 mg a.e./L for fathead minnow chronic exposure tests that lasted from 7 to 35 days. There was no obvious correlation with exposure time and NOEC. Since only one species was tested, an estimate of the chronic NOEC was made from the acute LC50 for cutthroat trout and the acute to chronic toxicity ratio. At the projected maximum use rate of 0.3 to 0.5 mg a.e./L, Hydrothol® 191 will chronically impact members of this segment of the biota. Due to the degree of uncertainty in the EEC value, the level of concern cannot be considered to be less than one in this case. In irrigation canals, chronic exposure does not occur because once the herbicide plume has passed, the EEC is essentially zero. Chronic field studies have not been conducted with Hydrothol® 191. However, the 1999 label indicates, that treatment rates of 1.0 mg a.e./L should not significantly impact the biota. This recommendation is supported by the experimental use of Hydrothol® 191 at 0.2 mg a.e./L during the 1999 season. However, *since Hydrothol® has the potential to be chronically adverse at concentrations in the 0.3 to 0.5 mg a.e./L range, use of Hydrothol® 191 at concentrations that exceed 0.2 mg a.e./L cannot be recommended* (Appendix D: Vol. 2, Sect. 4, p.10).

A field study in hard water indicates that Hydrothol® 191 is extremely toxic to a variety of fish in irrigation canals treated for 120 hours at 0.5 mg/L Hydrothol® 191. However, a 1999 treatment of Lake Steilacoom at 0.2 mg a.e./L for control of algae did not produce any obvious fish-kill (Appendix D: Vol. 2, Sect. 4, p.47).

Laboratory exposure of Chinook salmon at field rates of (0.2 mg a.e./L) indicates that Hydrothol® 191 interferes with the parr to smolt metamorphosis. It has been shown that after exposure to Hydrothol® 191 in freshwater, salmon smolts may not survive a seawater challenge (Appendix D: Vol. 2, Sect. 4, p.47).

Invertebrates Hydrothol® 191 has a high acute toxicity to free-swimming invertebrates. The LC50s range from 0.080 mg/L a.e. for *Daphnia magna* to 0.37 mg/L a.e. for the rotifer. At the projected maximum use rate of 5.0 mg a.e./L, Hydrothol® 191 is likely to acutely impact members of this segment of the biota. However, testing of more species of free swimming biota would lend greater confidence to risk assessment dealing with this segment of the biota. Since the maximum use rate of Hydrothol® 191 is 5.0 mg a.e./L even the least sensitive species of invertebrate within the biota will suffer adverse acute impact from the effects of Hydrothol®. For example, the risk quotient is significantly above the acute level of concern (0.1 for typical species) for all species tested. $RQ = 1.4 \text{ ppm a.e.} / 0.080 \text{ ppm a.e.} = \sim 18$. Field studies have not been conducted with free-swimming invertebrates exposed to Hydrothol® 191. Nevertheless, modeling studies indicate that treatment of canals with concentrations of Hydrothol® 191 as low as 0.2 mg a.e./L for 120 hours to control algae results in 20% of the resident invertebrate species being affected for 10 to 50 miles down stream (Appendix D: Sect. 4, pp.9-10).

Only two species of free-swimming invertebrates have been tested with Hydrothol® 191 for chronic toxicity. The experimental chronic toxicity (NOEC) is 0.016 mg a.e./L for *Daphnia magna* and <0.005 mg a.e./L for *Ceriodaphnia dubia*. At a projected maximum use rate of 0.3 to 0.5 mg a.e./L, Hydrothol® 191 will probably not chronically impact these Daphnid species (free-swimming invertebrate). No field studies were conducted that verify or deny the low chronic risk associated with Hydrothol® 191 against this segment of the biota. (Appendix D: Sect. 4, p.9).

Benthic Invertebrates Hydrothol® 191 has a high acute toxicity to benthic (sediment dwelling) invertebrates. For environmentally relevant species, the toxicity ranges from an LC50 of 0.12 mg a.e./L for the mayfly (*Hexagenia* sp.) to 1.6 mg a.e./L the northern crayfish (*Orconectes virilis*); some marine and estuarine species exhibit similar LC50s from 0.022 mg a.e./L for the grass shrimp to as high as 6.2 mg a.e./L for the fiddler crab. At a projected maximum use rate of 5.0 mg a.e./L, Hydrothol® 191 will acutely impact members of this segment of the biota. Field studies have not been conducted with these sediment species. However, since typical endofall sediment concentrations are 0.25 to 2 mg/L for a short period of time after application, the acute risk quotient may still exceed the level of concern (0.1) from this sediment exposure source. Therefore, exposure to sediment, or water (overlying, associated or pore) containing these concentrations of Hydrothol® 191 is likely to produce significant mortality or other adverse impact on this segment of the biota (Appendix D: Sect. 4, p.10).

Predicted chronic NOECs for Hydrothol® are used for benthic (sediment) invertebrates to predict risk since no laboratory studies were conducted. Use of Hydrothol® 191 at the maximum projected rate will not chronically impact the benthic biota. However, if concentrations found for 28 days in the sediment are considered (0.25 mg a.e./Kg) as representative of the EEC, the chronic level of concern would be exceeded and the sediment biota would be at risk. Although no field studies were conducted to verify the accuracy of this risk assessment, there is no reason to assume that predicted values for Hydrothol® for the chronic NOEC should follow a different acute to chronic toxicity ratio rules than for Aquathol® K, disodium endofall salt or endofall acid (Appendix D: Vol. 2, Sect. 4, p.10).

Endangered and Threatened Species Sensitive, endangered or threatened species of aquatic animals that may need protection through mediation include several salmon and trout species, thirteen rockfish species, two species of dace, two species of herring, and seven species of amphibian. Other species may also be sensitive, endangered, or threatened.

Water, Land and Shoreline Use

Swimming Lunchick (1994) conducted an exposure assessment to evaluate swimmers' exposure to endothall treated water. The exposure assessment was conducted according to EPA's standard operating procedures for swimmer exposure in treated water (Appendix D: Vol. 2, Sect. 5, p. 11).

The assessment included a maximum endothall use rate of 5 ppm concentration in the water and doses were calculated for persons 6 and 10 years of age and a 70 Kg adult. The exposure factors used in the assessment included body weights, body skin surface area, inhalation volume, an exposure time of 0.5 hr/day, exposure to water and through the mouth of 5000 ml/hr and the amount of water swallowed of 50 ml/hr. In addition, the endothall constants of 5 ppm use rate, skin permeability coefficient of 10⁻⁴ cm skin/hr, octanol/water partition coefficient of 0.008 and endothall vapor pressure of 3.92 x 10⁻⁵ mm Hg were included in the calculations of the daily oral, dermal, mouth (buccal/sublingual) and ear canal exposures (Appendix D: vol. 2, Sect. 5, p. 11).

Lunchick (1994) calculated that the daily total dose to a person swimming in water containing 5 ppm endothall was extremely low and did not present an acute toxicity risk. The results of the assessment review revealed that 95-97% of the swimmers daily dose of endothall was due to ingestion or swallowing the treated water. The remainder of endothall exposure was 2-3% and 1% to the skin and inhalation routes, respectively. The total calculated endothall daily doses for the 6 and 10 year olds and a 70 kg adult were 5.9, 3.7 and 1.9 ug/kg/day, respectively (Appendix D: Vol. 2, Sect. 5, p. 11).

Skin Irritation Findings from the Hydrothol® 191 skin irritation study demonstrated the product to have a severe degree of irritation. It is noted that signs of severe dermal edema and erythema and necrosis were observed at 30-60 minutes, 24, 48 and 72 hours following dermal application. The primary skin irritation score was 7.83/8.0, classing Hydrothol® 191 as an EPA FIFRA Toxicity Category I skin irritant (Appendix D: Vol. 2, Sect. 5, p.5).

Note: Washington State Department of Health (DOH) made the following comments to Appendix D, Vol. 2, Sect. 5 of the risk assessment.

{Risk Assessment} conclusion on page 12 implies that no eye irritation is expected from any endothall products at treatment concentrations. This conclusion is apparently based solely on tests of endothall technical and Aquathol as no data on Hydrothol 191 are presented. It is possible that Hydrothol is comparatively more irritating to eyes. Given the notable difference between Hydrothol and Aquathol or endothall technical in the skin sensitivity testing, Hydrothol may be much more irritating. If this is the case, regulation of all endothall products on the basis of Aquathol data may not be protective of public health. Please consider a swimming restriction of 24 hours for Hydrothol products, again to protect against irritant symptoms. This restriction could be dropped if data are submitted which demonstrate the lack of eye irritation at treatment concentrations (Morrissey 2000).

Agriculture At typical use rate concentrations, irrigation or flooding of crops with water that has been treated with Hydrothol® 191 should not cause significant damage. Endothall is unlikely to bioaccumulate in livestock or fish. Since the mode of application of Aquathol® and Hydrothol® is typically by subsurface injection or sinking granules, drift is likely to be minimal (Appendix D: vol. 2, Sect. 4, p. 38).

4. Mitigation

Use Restrictions Hydrothol® 191 will have an acute or chronic impact on the biota when applied at concentrations recommended on the label. Field data indicate that Hydrothol® 191 cannot be used to control weeds at concentrations higher than 0.5 mg a.e./L without significant fish-kill. Hydrothol® 191 should not be used to control species of weeds that are not specified in the label. Hydrothol® 191 has been recommended by some for the control of toxic blue-green algae at concentrations that may not harm green algae. Insufficient field data have been collected to know with certainty what concentrations of Hydrothol® can be maximally used and not harm resident biota. However, enough data has been collected to show that Hydrothol® at concentrations higher than 0.2 to 0.5 mg a.e./L can harm fish and possibly free-swimming and benthic biota. To mitigate the effects of the use of Hydrothol® 191, the lowest concentration that will achieve the desired control of algae should be used. *Currently, a safe treatment rate of higher than 0.2 mg a.e./L cannot be recommended without potential for acute and chronic adverse impact. The exposure period should be as low as possible (high flow-rates in canals), the minimum area possible should be treated; treatments of water bodies that contain hard water should be avoided; and treatments should occur from the shoreline outward to allow for the possible avoidance of Hydrothol® by free-swimming fish* (Appendix D: Vol. 2, Sect. 4, pp. 11, 72).

Swimming/skiing Current labels for Hydrothol® 191 Aquatic Algaecide and Herbicide (EPA Reg. No. 4581-174) and Hydrothol® 191 Granular Aquatic Algaecide and Herbicide (EPA Reg. No. 4581-172) do not contain swimming restrictions; however, a 24 hour swimming restriction is recommended in treated areas to protect against irritant symptoms. All General Mitigation posting requirements apply. In addition, informational buoys should be placed around the treatment area. Swimming outside the treatment area is permitted.

Boating A 24 hour restriction is recommended for boaters (excluding licensed applicators) entering the area of treatment for protection against eye irritation due to drift or dust.

Drinking/Domestic Uses Hydrothol® 191 Aquatic Algaecide and Herbicide (EPA Reg. No. 4581-174) and Hydrothol® 191 Granular Aquatic Algaecide and Herbicide (EPA Reg. No. 4581-172) labels restrict the use of water from treated areas for watering livestock, preparing agricultural sprays for food crops, irrigation or domestic purposes up to 25 days after application, depending on the concentration used. Please refer to the label.

Fisheries Exposure to fisheries should be avoided. Current labels warn that fish may be killed at dosages in excess of 0.3 ppm. *Currently, a safe treatment rate of higher than 0.2 mg a.e./L cannot be recommended without potential for acute and chronic adverse impact. The exposure period should be as low as possible (high flow-rates in canals), the minimum area possible should be treated; treatments of water bodies that contain hard water should be avoided; and treatments should occur from the shoreline outward to allow for the possible avoidance of Hydrothol® by free-swimming fish* (Appendix D: Vol. 2, Sect. 4, pp. 11, 72).

Endangered Species Levels of Hydrothol® 191 that would be found in the environment due to typical treatment practices may interfere with the salmon smoltification process resulting in death when smolts migrate from freshwater to saltwater. Extra caution should be taken when dealing with endangered species allowing for a level of concern of <0.05 rather than the more typical value of <0.1 for non endangered species. Restrictions on season of application are warranted to protect sensitive salmon smolts from the effects of endothall products; similar restrictions may be applied to protect fish and

fisheries and prevent water use restrictions during the height of the recreational and commercial fishing seasons (Appendix D, Vol. 2, Sect. 4, p. 67).

Fish Consumption Hydrothol® 191 current labels restrict use of fish from treated areas for food or feed within 3 days of treatment.

Wetlands Lakes with outflows to estuaries should be treated only at low flows or with a buffer around the outlet.

Post-treatment Monitoring If treatment occurs near domestic water sources, post treatment monitoring may be desirable.

References

Please refer to citations in Appendix D: Compliance Services International, 2000. *Supplemental Environmental Impact Statement Assessments of Aquatic Herbicides*, Volume 2: Endothall. 275 pages.

Morrissey, Barbara, 2000. Letter dated June 22, 2000, to Kathleen Emmett, Washington State Department of Ecology from Barbara Morrissey, Toxicologist, Washington State Department of Health.

L. Fluridone

1. Registration

DowElanco registers Fluridone under the trade name of SONAR.

Typical Use Fluridone takes 30 to 90 days, under optimum conditions, to completely kill target plants such as elodea and Eurasian watermilfoil. For best results, Sonar, a systemic herbicide, should be applied just before or just after plants begin to grow. Sonar should not be applied in situations where heavy rains may dilute treated water or where there is rapid water movement unless applied to an area 5 acres or greater. Sonar may not be effective when used to treat shorelines or small areas (spot treatments).

2. Description

Fluridone was originally developed as a terrestrial herbicide for use in cotton (Webster et al. 1977, Wills 1977, Banks and Merkle 1978). In numerous studies of fluridone and terrestrial soils, researchers have investigated adsorption and desorption, persistence, retention and release, chemical and physical properties, and effect of soil pH on fluridone activity (Loh et al. 1978; Parka et al. 1978; Banks et al. 1979; Banks and Merkle 1979; Shea and Weber 1980, 1983; Weber 1980; Schroeder and Banks 1986a, 1986b; Weber et al. 1986; McCloskey and Bayer 1987).

Fluridone is a systemic herbicide that moves from submersed foliage to roots or emerged foliage (Marquis et al. 1981, Westerdahl and Getsinger 1988). A plant's susceptibility to fluridone is associated with its uptake rate and rate of translocation. Fluridone interferes with the synthesis of RNA, proteins, and carotenoid pigments and thereby affects photosynthesis (Bartels et al. 1978, Berard et al. 1978, Wells et al. 1986). Visible herbicidal effects on treated plants include pink or chlorotic growing points within seven to 10 days after application.

Toxicity Fluridone is not teratogenic, not mutagenic, and is not listed or considered to be carcinogenic. There have been no reports of significant exposure to fluridone. In case of a large spill, material should be prevented from flowing into streams, ponds, or lakes, or onto adjacent land.

3. Environmental and Human Health Impacts

Earth

Soils and Topography In general, impacts to soils should be slight with the aquatic use of fluridone. Fluridone degradation and persistence in hydrosols remains a concern because impacts to vegetation may occur long after treatment. Wetland or "unique" species in and along the perimeter of the treatment area could be killed.

Sediments In early field trials with liquid fluridone, investigators found that fluridone residues reached a maximum in hydrosol 14 days after treatment (Grant et al. 1979). No detectable residue (at a test sensitivity of 0.010 ppm in hydrosol) was observed in hydrosol after 62 days. In a later study of two ponds in Indiana, the residue pattern was similar (using two different methods) in both ponds, with no detectable residue remaining 56 days after treatment (West and Parka 1981). Muir and Grift (1982)

found that the half-life of fluridone in artificial ponds under field conditions was 17 weeks and 12 months under laboratory conditions. No detectable residues were observed in hydrosol in ponds after one year in ponds and lakes in three geographic regions in the US and in Panama (West et al. 1979). Thus, time for fluridone to reach no detectable residues in hydrosol in the field ranged from eight weeks to 12 months.

In the laboratory, Marquis et al. (1982) studied degradation of fluridone in sandy and silt loam submersed soils under controlled conditions. The laboratory conditions eliminated photolysis and plant uptake, the normal mechanisms for fluridone removal from hydrosol. Additionally, laboratory conditions minimized hydrosol adsorption, but did allow microbial metabolism. Investigators found that fluridone persisted slightly longer in the water above the silt loam than in the water above the sandy loam. Thirty percent of the parent compound remained in the soils after 12 months under artificial conditions.

Invasive non-native species such as Eurasian watermilfoil can reduce current speed in flowing water. Reduced current speed may in turn increase siltation, which can change the shape and/or composition of river or lake bottoms. Removal of undesired aquatic vegetation can result in increased flow rates and decreased siltation.

In conclusion, fluridone degradation and persistence in hydrosols remains a concern. Since fluridone can be absorbed from the hydrosol by roots, and since fluridone persists in the hydrosol for extensive periods of time, impacts to vegetation may occur long after treatment.

Air

Air Quality Adverse impacts to air quality are expected to be minor, such as a small amount of exhaust emissions associated with the use of application equipment. No aerial drift or overspray is expected since herbicide application is usually performed with subsurface applicator devices and volatilization of fluridone is insignificant.

Drift of fluridone into non-treatment areas may occur depending on the chemical formulation and suspending agent used. Removal of native species may allow infestation by noxious species such as Eurasian watermilfoil.

Water

Surface Water Fluridone has a water solubility of 12 ppm; water solubility can strongly influence the environmental fate and persistence of a herbicide (Westerdahl and Getsinger 1988). There are no label restrictions against drinking, swimming, or fishing in water treated with fluridone (EPA 1986). EPA has established a drinking water standard for fluridone of 0.15 ppm., and the EPA registration label recommends waiting 7 to 30 days before using treated water for irrigation.

Numerous investigators have measured the half-life of fluridone in surface water with a range of results. Hall et al. (1984) and Elanco Company stated that the apparent half-life of fluridone in water is 14 days or less. Fluridone aqueous half-lives ranged from 5 to 60 days in a study by West et al. (1983), from 4 to 7 days in a Canadian pond study (Muir et al. 1980), and from 2 to 3.5 days in another Canadian pond study (Muir and Grift 1982). Weed Science Society of America (1983) stated that fluridone has a half-life of 21 days in water when used for control of aquatic vegetation.

In a further study, West and Parka (1981) observed in two ponds using two methods of detection that the rate of fluridone dissipation from water was similar in both ponds. The half-lives of fluridone were 21 and 26 days after surface application and application along the pond bottom. They concluded that the method of applying fluridone to the pond did not appear to affect herbicide dissipation from the aquatic environment.

Finally, Grant et al. (1979) observed that fluridone began to dissipate from the water in 3 to 14 days after treatment, while Kamarianos et al. (1989) observed that fluridone levels in a Greek pond populated with carp decreased to below detection limits after 60 days. In the Greek study, fluridone decreased in the water at a high rate during the first days after application, and no fluridone was detected after two months, results similar to Langeland and Warner (1986).

The primary fate process affecting fluridone persistence in aquatic environments is photolysis (West et al. 1983). Fluridone is stable to oxidation and hydrolysis (McCowen et al. 1979), and volatilization is not expected to be significant. A photolysis half-life of 5.8 days for fluridone was observed in flasks containing pond water (Muir and Grift 1982). In another study, Saunders and Mosier (1983) observed that photochemical half-lives of fluridone ranged from 22 to 55 hours and were only slightly dependent on initial fluridone concentrations. Recent studies indicate that radiation between 297 and 325 nm is primarily responsible for the photodegradation of fluridone (Mossler et al. 1989). The half-life of fluridone is 26 hours at these wavelengths.

Fluridone is not as effective in flowing waters as in impounded waters because it is a slow-acting herbicide. Accordingly, some researchers have been studying various methods to prolong its dissipation. Van and Steward (1985) found that use of large diameter fibers for controlled delivery of fluridone in moving water could extend its use. In their study, fluridone release lasted over 40 to 50 days (no detectable fluridone levels were determined after 42 days using 0.8 and 1.2 mm fibers). Dunn et al. (1988) packaged fluridone in fibers that became trapped in aquatic plants. They controlled the release rate of fluridone by adjusting concentration in the fibers, and achieved fairly constant release rates from several days to four months.

Fluridone can persist for months when applied in the fall. The decreased temperatures and low light levels slow its dissipation from water. This has resulted in using fluridone for fall applications in the Midwest where lakes freeze (Hamel, personal communication, 2000).

Adverse impacts to surface water quality may occur after fluridone treatment of aquatic vegetation. After death, plants decompose which may create a short-term biological oxygen demand and a longer-term increase in organic sediments. Problems with decreased dissolved oxygen levels are not expected with fluridone because it is such a slow-acting herbicide with effects occurring over a long period. Field studies in Washington have shown little to no dissolved oxygen sag when fluridone was applied to the whole lake for noxious weed control (Hamel, personal communication, 2000).

Increased concentrations of organic and inorganic phosphorus may occur in the water column during plant decomposition following fluridone treatment. Phosphorus is often a limiting nutrient in aquatic plant growth, therefore use of fluridone may result in rapid phytoplankton growth. Dense alga blooms were observed in Swofford Pond, Lewis County, after treatment with fluridone. Another problem observed in Swofford Pond following treatment was the formation of a dense berm of decomposing plants at one end of the lake. The berm was a source of nutrients to the lake and a navigation hazard. However, this situation seemed limited to Swofford Pond. Berms have not been observed in other lakes treated with fluridone (Hamel, personal communication, 2000).

Drift of fluridone into non-treatment areas may occur depending on the chemical formulation and suspending agent used, and on currents in the treatment area. However, fluridone is usually not used in areas with currents because of the long uptake time needed for the chemical to be effective. (See also sections on Public Water Supply and Habitat.) Fluridone is not recommended for spot treatment use

Ground Water No direct ground water contamination issue is associated with the application of fluridone to aquatic sites (EPA 1986), though this issue may warrant further study because fluridone has been included on an EPA list of chemicals suspected to leach. There are no label restrictions against drinking, swimming, or fishing in water treated with fluridone. Primarily photolysis, by biodegradation, and least significantly by volatilization (Westerdahl and Getsinger 1988) degrade Fluridone.

Public Water Supplies In summary, no adverse effects are anticipated due to exposure to fluridone under the expected conditions of use (Appendix H). Drinking water must not exceed 0.15 parts per million to meet EPA's drinking water tolerance (Wisconsin DNR 1990), and the label recommends waiting from 7 to 30 days before using fluridone treated water for irrigation. The EPA label does not restrict use of fluridone-treated water for swimming or domestic purposes, but does contain a restriction against use of fluridone within 1/4 mile of any potable water intake (Appendix H, Fluridone Human Health Risk Assessment). However, when treating Eurasian watermilfoil at rates of 20 ppb or less, this restriction does not apply.

Reviewing available toxicology data then calculating an acceptable dose for each formulation assessed the risk to human health due to the use of fluridone. Next, a maximum acceptable concentration (MAC) in the water was determined based on expected human ingestion rates of water or aquatic organisms. The MAC was then compared to the estimated environmental concentration (EEC) (the concentration in a waterbody calculated from herbicide application rates and persistence data). If the EEC is less than the MAC, no increased risk to human health is expected (Appendix H, Fluridone Human Health Risk Assessment).

Significant routes by which the general public can be exposed to aquatic herbicides are:

- 1) Using the waterbody as a drinking water source (ingestion),
- 2) Swimming (incidental ingestion and dermal exposure),
- 3) Eating aquatic organisms (ingestion).

The acceptable dose (dose at which no adverse effects are expected to occur) for fluridone was calculated based on available toxicology data and on EPA regulations. This concentration, which was determined for each route of exposure, would be expected to cause no adverse effects to human health. The calculation of an acceptable dose assumes that the herbicide is not carcinogenic, and fluridone has been determined by EPA not to cause cancer.

For water ingestion, two intake rate scenarios were used, a worst-case analysis assuming the treated water was used as the drinking water supply, and a more likely exposure scenario assuming incidental water ingestion while swimming. The incidental ingestion scenario is still conservative because it was assumed that people were exposed daily for a prolonged period of time (chronic exposure) to initial herbicide concentrations. Potential exposures would actually be much more limited when applications of herbicides only occurred once per year, and degradation half-lives reported in field studies range from 5 to 60 days for fluridone.

Estimated initial water concentrations did not exceed either the water supply MAC or the incidental ingestion MAC for adults or children. Also, estimated initial concentrations did not exceed calculated

MACs for fluridone for the dermal exposure route and the ingestion of aquatic organisms. For dermal exposure, the model used to calculate a MAC was based on the assumption that contaminants are carried through the skin as a solute in water. Thus, the flux rate of water across the skin boundary was assumed to be the factor controlling contaminant absorption rate. For ingestion of aquatic organisms, the contaminant intake rate was calculated from a daily fish ingestion rate (6.5 grams/day) multiplied by a bioconcentration factor for accumulation of the contaminant in fish tissue.

In addition to potential risks from systemic absorption of the herbicides, there is a potential for effects from direct contact of herbicides with skin and eyes. Fluridone is not irritating to the skin, and only minor effects were noted after application of undiluted fluridone to the eyes of rabbits. Thus, no adverse effects are expected from contact with dilute solutions.

An issue associated with the use of fluridone concerns a potential photolytic breakdown product. N-methyl formamide (NMF) is a potential teratogen, fetotoxin, hepatotoxin, and cytotoxin. NMF was first observed in laboratory photolytic studies using distilled water and lake water (Saunders and Mosier 1983). However, NMF was not observed in field studies conducted outdoors in artificial ponds with radiolabelled fluridone (Berard and Rainey 1981 in Osborne et al. 1989) or in experimental ponds in Florida at a detection limit of 2 ppb (Osborne et al. 1989).

Although NMF has never been observed as a breakdown product under natural conditions, its potential presence was a concern in the 1992 SEIS. Therefore, worst case calculations were performed on its potential to affect human health (Appendix A3). In summary, the safety factors for NMF exposure through drinking water and through skin absorption are very high, both under a worst case scenario (30,303 X and 1,111,111 X, respectively) and under more realistic conditions (>149,254 X and >5,555,555 X). Under worst case conditions, a person would need to drink 15,852 gallons of treated drinking water per day to reach the NOEL, or greater than 78,077 gallons per day under realistic case conditions. For incidental ingestion, a person would have to swim in fluridone treated water for 1,014 years under worst case conditions and for >5,070 years under realistic case conditions in order to be exposed to equal the NOEL.

It has been concluded that the use of fluridone according to label instructions does not pose any effect to human health. These are large margins of safety, and the amount of water a person would need to drink or the time a person would need to swim to reach the NOEL is very unrealistic (Appendix H, Fluridone Human Health Risk Assessment).

Plants

Habitat Fluridone is an herbicide that is taken up by both shoot and root tissue of submersed vascular aquatic plants and moved to other parts of the plant within the vascular system (McCowen et al. 1979, Marquis et al. 1981). Translocation rate and direction (i.e. root to shoot or shoot to root) appear to be somewhat species dependent. Noticeable "dying off" or decrease in biomass of vegetation treated with fluridone begins approximately 8-16 days after treatment (Hall et al. 1984).

Fluridone interferes with the synthesis of RNA, proteins, and carotenoid pigments in aquatic plants causing death by a form of sunburn (carotenoid pigments protect chlorophyll from ultraviolet light) (Bartels and Watson 1978, Berard et al. 1978). Anderson (1981) concluded that treated American pondweed or sago pondweed need exposure to sufficient light for fluridone to work effectively, and that turbid water may reduce fluridone's effectiveness on these species. Fluridone affects a variety of aquatic

plants. A list of species susceptible to fluridone at an application rate of 0.1 ppm follows (Parka et al. 1978, Arnold 1979):

Hydrilla	<i>Hydrilla verticillata</i>
American Elodea	<i>Elodea canadensis</i>
Fanwort	<i>Cabomba caroliniana</i>
Eurasian watermilfoil	<i>Myriophyllum spicatum</i>
Coontail	<i>Ceratophyllum demersum</i>
Illinois pondweed	<i>Potamogeton illinoensis</i>
Southern naiad	<i>Najas guadalupensis</i>
Parrotfeather	<i>Myriophyllum brasiliensis</i>
Common bladderwort	<i>Utricularia spp.</i>
Water celery	<i>Vallisneria spp.</i>
Arrowhead	<i>Sagittaria spp.</i>
Cattail	<i>Typha spp.</i>
Spatterdock	<i>Nuphar luteum</i>
Reed canarygrass	<i>Phalaris arundinacea</i>

In addition to the plants listed above, Elanco Products Company lists the following aquatic plants as susceptible to fluridone: creeping waterprimrose (*Ludwigia peploides*), waterpurslane (*Ludwigia palustris*), bulrush (*Scirpus spp.*), rush (*Juncus spp.*), smartweed (*Polygonum spp.*), spikerush (*Eleocharis spp.*), and waterlily (*Nymphaea spp.*).

Grant et al. (1979) reported on the time required for control of similar species at various fluridone concentrations, and McCowen et al. (1979) evaluated fluridone at various rates in 265-liter containers on several aquatic plants.

The variable efficacy of fluridone reported in the literature may be partly explained by a recent study. Spencer and Ksander (1989) exposed hydrilla to fluridone at various concentrations for 1, 3, and 5 weeks. Plant recovery was directly related to the concentration of active iron (Fe^{2+}) in the plant at the time of treatment. Concentration of iron in the water during treatment did not reduce the phytotoxicity of fluridone. However, plants exposed to fluridone for three weeks did not recover even though the active iron concentration in the plant was high before treatment.

Parka et al. (1978) observed that fluridone did not appear to adversely affect desirable phytoplankton. Some reductions of less desirable phytoplankton such as *Anabaena* and *Anacystis* occurred after treatment at 0.3 and 0.1 ppm. In a study conducted in Greek ponds, a drastic reduction in phytoplankton species was observed shortly after fluridone application, and the population of *Cyanophyceae* (Cyanobacteria) disappeared after about two months (Kamarianos et al. 1989). The more desirable species such as diatoms increased significantly, especially epiphytic and benthic species.

In a study conducted in the laboratory, researchers concluded that fluridone may be toxic to alga growth and N_2 -fixation at concentrations between 0.5-10 $\mu g/l$ (Trevors and Vedelago 1985). Recovery from fluridone treatment was not apparent when *Scenedesmus quadricauda* was incubated for an extended period of time. It should be noted that actively growing cultures of *S. quadricauda* were relatively insensitive to fluridone compared to cultures exposed at the beginning of the bioassay.

Parka et al. (1978) reported that trees and shrubs growing in water were damaged by fluridone applications. When vegetation was growing on the bank, no phytotoxic symptoms were observed.

Impacts from release of nutrients during plant decomposition following fluridone treatment may include increased nutrient levels. Increased nutrient concentrations may result in increased alga blooms or in increased growth of other aquatic plants.

Due largely to the long contact time needed for the herbicide to work, fluridone may be carried away from the application area before it is effective. This problem has been addressed by loading fluridone in fibers to adjust the release rate (Dunn et al. 1988). Also, treated areas are usually a small portion of the lake at any one time, with the exception of whole-lake treatment for eradication of a non-native species such as Eurasian watermilfoil. (See also Sections on Threatened and Endangered Species, Aesthetics, and Agricultural Crops). Treatment of Eurasian watermilfoil with fluridone may allow native species to return after several years and may result in a greater diversity of species rather than a monoculture of watermilfoil.

Animals

Fluridone has a very low order to toxicity to zooplankton, benthos, fish, and wildlife, but may remain in fish tissue up to 120 days after treatment (Parka et al. 1978, McCowen et al. 1979, Arnold 1979, Grant et al. 1979). Acute and 90-day subacute toxicological results for technical grade fluridone indicate the following (Parka et al. 1978):

ORGANISM	ROUTE	CONCENTRATION	
<u>Daphnia</u>	Water	LC ₅₀	6.3 ppm
Rainbow Trout	Water	LC ₅₀	11.7 ppm
Bluegill	Water	LC ₅₀	14.3 ppm
Bobwhite Quail	Diet	LC ₅₀	ca 10,000 ppm
Mallard Duck	Diet	LC ₅₀	> 20,000 ppm
Mallard Duck Acute	Oral	LD ₅₀	> 2,000 mg/kg

No adverse effects were observed on crayfish, bass, bluegill, catfish, long-neck soft-shell turtles, frogs, water snakes, and waterfowl from the use of 0.1 to 1.0 ppm fluridone during field experiments (Arnold 1979, McCowen 1979). Zooplankton were reduced slightly when 1.0 ppm was applied, but populations quickly recovered. Total numbers of benthic organisms did not change significantly at 0.3 ppm; however 1.0 ppm did affect total numbers (Parka et al. 1978).

Similar observations have been made with fluridone use in other parts of the world. Investigators of Gatun Lake, Panama, concluded that total numbers and community structure of zooplankton, phytoplankton, and benthic organisms did not vary significantly during field tests of fluridone (Theriot et al. 1979). Kamarianos et al. (1989) concluded that no detrimental effects occurred in fish productive aquatic ecosystems (Greek ponds) treated with fluridone.

The uptake rate and clearance of fluridone by aquatic organisms is very low. Rainbow trout had a bioconcentration factor of 91 estimated by a pharmacokinetic model, while *Chironomus tentans* (4th instar) had an estimated bioconcentration factor of 128 (Muir et al. 1982). The relatively high concentration of fluridone in the sediments during these experiments did not appear to have serious adverse effects on chironomid larvae.

Parka et al. (1978) and Arnold (1979) reported that fluridone did not accumulate in fish. It was observed in bodies of bluegills 15 days after treatment, but the amount in the head or body did not exceed the

concentration in the water. Grant et al. (1979) showed that channel catfish contained a low fluridone residue (0.015 ppm) 120 days after treatment of ponds, but no fluridone residue was detected in largemouth bass or bluegill fish. Fluridone did not bioconcentrate in any of the fish species. In laboratory tests using mosquito fish (*Gambusia affinis*), McCowen et al. (1979) observed that they survived and produced young at all rates of fluridone treatment.

In a recent study, Hamelink et al. (1986) reported that fathead minnows were not affected by continuous exposure to fluridone of 0.48 mg/l or less over their life cycle. The researchers did not observe any effects when daphnids, amphipods, or midge larvae were continuously exposed to concentrations of fluridone (0.2 mg/l or less for 32 days, 0.6 mg/l or less for 60 days, or 0.6 mg/l or less for 30 days, respectively). They determined that the acute median lethal concentrations of fluridone were 4.3 mg/l for invertebrates and 10.4 mg/l for fish. In the same study, growth and survival of channel catfish were not negatively affected by continuous exposure to fluridone concentrations of 0.5 mg/l or less for 60 days after hatching. They also observed that channel catfish accumulated fluridone concentrations 2 to 9 times greater than concentrations in water, for a bioconcentration factor of 2 to 9.

In concluding remarks, Hamelink et al. (1986) stated that a favorable safety margin exists between fluridone concentrations that affect non-target organisms and concentrations needed to control weeds. They observed that the recommended application of 1 lb/acre of fluridone to a pond with an average depth of 3 ft provides a theoretical concentration of 0.1 mg/l; therefore an initial fluridone concentration of 0.1 mg/l or less is recommended to control weeds in ponds. Consequently, fluridone is not expected to have adverse effects on the species tested or on similar non-target aquatic organisms.

Estimated environmental concentrations (EEC) for fluridone expected to occur in the water after applications at the recommended rate are 0.13 ppm (Final Acute Value, Final Residue Value, and Criterion Maximum Concentration), and 0.08 ppm (Final-Chronic Value and Criterion Continuous Concentration) (Aquatic Environmental Risk Assessment, Appendix I). None of the criteria values are exceeded for fluridone; therefore it should be possible to use this herbicide without significant risk to 95% or more of aquatic animal species. However, up to 5% (statistically) of aquatic species could be impacted adversely. Economically important and endangered/threatened species are expected to be protected at the forecast herbicide application rates and estimated exposure concentrations (See also Appendix I).

Threatened and Endangered Species Treatment with herbicides has the potential to affect submersed and emersed plant species federally listed as rare, threatened or endangered. These species may be aquatic or may occur along the banks of waterways. Applications for short-term modifications to water quality standards for each specific site should include a review of the rare, threatened, or endangered species listed by US Fish and Wildlife, and of "proposed sensitive" plants and animals listed by Washington State National Heritage Data System.

The bioconcentration factor (BCF) of fluridone in fish ranges from 0.9 to 3.7 (Elanco 1985) and from 1.6 to 15.5 (West et al. 1983) (although Muir et al. (1982) report a BCF of 91 for rainbow trout and 128 for *Chironomus tentans*). Hamelink et al. (1986) observed that the BCF for fluridone in catfish ranged from 2 to 9. A value of 100 is usually regarded as a significant factor. Given there is a very low probability that fluridone will bioaccumulate or biomagnify in fish, the need for concern for bald eagles and other threatened or endangered predators of fish in treated areas is also low.

Water, Land and Shoreline Use

Navigation should improve after treatment of large areas of vegetation. In field tests, McCowen et al. (1979) found that aquatic plants gradually sank to the bottom two to four weeks after treatment thereby increasing the amount of open water. Open water will result in decreased impacts to electric and gas motors used to propel boats. Boat lanes that are often maintained for navigation through dense aquatic vegetation would not be necessary after treatment.

Swimming There are no swimming restrictions associated with fluridone treatment. Treatment is expected to improve swimming conditions when used to remove dense plant populations from areas used for swimming. Increased open water areas could improve other recreational activities such as water skiing and boating (Hamel, Ecology, pers. comm.). In-depth health-risk analysis indicates use of Sonar at label application rates would not result in adverse impacts on human health. Analysis included methods for assessing potential health risks associated with fluridone, and those associated with N-methyl formamide (NMF, which is a potential breakdown product of fluridone). EPA has determined that fluridone does not cause cancer (Appendix H).

Fishing could be both positively and negatively impacted. Navigation to and from fishing areas would be improved after fluridone treatment. Also, fish are not significantly affected at treatment concentrations. Habitat for some fish species could be improved with treatment (e.g., planted trout spp.), whereas habitat for other species might be reduced with treatment (e.g., bluegill, crappie, yellow perch).

Agriculture Irrigation with fluridone treated water may result in injury to the irrigated vegetation. As a special precaution notice on the label, Elanco recommends informing those who irrigate from fluridone treated areas not to irrigate established tree crops for seven days after treatment; not to irrigate established row crops, turf, or plants for 14 days from canals, lakes, and reservoirs, and 30 days from ponds and static canals; and not to irrigate newly seeded crops, seedbeds, or areas to be planted (including overseeded golf course greens) for 14 days from lakes and reservoirs, and 30 days from canals, static canals, and ponds.

A few investigators have studied effects of fluridone on agricultural crops. Corn was less sensitive to fluridone than wheat, although the carotenoid content of wheat and corn dropped dramatically when these plants were treated with fluridone (Devlin et al. 1978). When grown in nutrient solutions with fluridone, soybean, corn, and rice seedlings developed phytotoxic symptoms after 3-5 days exposure (Berard et al. 1978). In comparative phytotoxicity studies, Banks (1978) indicated that resistance of selected crop species to fluridone were cotton > peanuts > soybeans > corn > wheat = grain sorghum = oats = rye.

Fluridone was originally developed for weed control in cotton; it showed broad spectrum herbicidal activity (Webster et al. 1979, Miller and Carter 1983). Cotton showed tolerance for fluridone, which was believed to be associated with reduced root uptake and translocation than found in other species (Albritton and Parka 1978, Berard et al. 1978, Raffi and Ashton 1979). Fluridone also caused inhibition of metabolic processes (Raffi et al. 1979).

When used for terrestrial weed control in field trials, fluridone provided effective control of yellow nutsedge (*Cyperus esculentus*), purple nutsedge (*Cyperus rotundus*), rhizome johnsongrass (*Sorghum halepense*), and common bermudagrass (*Cyanodon dactylon*) (Webster et al. 1979). Banks (1983) observed that in cotton, fluridone was the most effective herbicide in reducing tuber and shoot populations of yellow nutsedge, a troublesome and costly weed in southeast US. In another terrestrial

study, fluridone was effective in control of the weeds puncturevine (*Tribulus terrestris*), kochia (*Kochia scoparia*), and barnyard grass (*Echinochloa crusgalli*); less effective on poison suckleya (*Suckleya suckleyana*), and lanceleaf sage (*Salvia reflexa*); and not effective on sunflower (*Helianthus annuus*) or cocklebur (*Xanthium pennsylvanicum*) (Crutchfield and Wiese 1979).

When used as a terrestrial herbicide, fluridone was persistent. Shea (1981) showed that soil organic matter levels strongly influenced the activity and residual phytotoxicity of fluridone. Fluridone residues eight months after treatment reduced growth of several crops and weeds by 75% or more (Miller and Carter 1983). Fluridone's persistence in soil restricted its acceptance and use in western irrigated agriculture.

In aquatic studies, the time to no detectable residue level for the aqueous form of fluridone is between 2 months to 12 months (levels approached zero 64-69 days after treatment) (Langeland and Warner 1986). Based upon the moderate persistence of fluridone observed in this study, caution should be used when applying fluridone to enclosed irrigation source ponds until more information is available on phytotoxicity of crops to low concentrations of fluridone in irrigation water.

4. Mitigation

The major potential impacts associated with the use of fluridone included loss of non-target species, persistence in sediment, and water quality impacts associated with plant decomposition. In addition to potential mitigation measures listed previously, permits allowing the use of fluridone should only be issued where treatment is considered necessary. Provisions for pre- and post monitoring of vegetation impacts and a mitigation plan were mandatory, but Ecology has extensive monitoring information now and this is no longer necessary. The label has been changed from stipulating that fluridone must not be used within 1/4 mile of an intake withdrawing water for domestic purposes to an allowance of 20 ppb over water intakes on the most recent label.

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M. Glyphosate

1. Registration

Glyphosate was first registered for use in 1974. Glyphosate is formulated as Rodeo® or Pondmaster® for use in aquatic sites, and as Roundup® for terrestrial use. In this review, we do not consider impacts of Roundup® (except on soils), but only the two aquatic formulations which do not include a surfactant. The Roundup® formulation includes the surfactant POEA (polyethoxylated tallow amine), which has demonstrated toxic effects on aquatic organisms.

Glyphosate is the active ingredient in Rodeo®, an herbicide manufactured by Monsanto and registered by EPA for aquatic use. Glyphosate is also the active ingredient in Roundup, a product that is not approved for use in aquatic environments due, at least in part, to potential adverse impacts to fish from the surfactant used in the Roundup formulation. This surfactant is not used in the Rodeo formulation. Glyphosate was evaluated for use in the Aquatic Plant Management Program in an addendum to the 1980 Environmental Impact Statement entitled *Aquatic Plant Management Through Herbicide Use*. Glyphosate was not used in aquatic systems in Washington in 1980 and was therefore not formally evaluated as part of the 1980 FEIS.

Typical Use Glyphosate effectively controls purple loosestrife, cattails, and floating-leaved plants such as water lilies, duck weed, and watershield. Glyphosate loses its effectiveness soon after coming into contact with water, therefore, Rodeo is not effective on submerged plants. Loss of effectiveness is heightened by sediment in the water. Other factors that may reduce effectiveness include rainfall within 6 hours after treatment, and wind, which may reduce the amount of active ingredient that reaches the target.

2. Description

Glyphosate (N-(phosphonomethyl)glycine) (trade name: Rodeo®) is a broad spectrum herbicide employed for the control of emerged aquatic grasses, broadleaf weeds, and brush (Westerdahl and Getsinger 1988). Glyphosate is applied to emerged foliage, but not to submersed or mostly submersed vegetation. The isopropylamine salt of glyphosate is used for aquatic plant control and is applied when plants are actively growing. The maximum water concentration is not specified although the recommended concentration is 0.2 mg/l. Recommended concentrations range, for emergent aquatic vegetation, from 5.3 to 8.8 l/ha as a broadcast spray, and from 0.75 to 1.5% hand sprayed. The mode of action of glyphosate is not definite, but biosynthesis of phenylalanine may be interrupted through repression of chorismatic acid, plant elongation may be inhibited, and photosynthesis may be disrupted. Glyphosate is rapidly absorbed and translocated throughout plant tissues to control the entire plant including leaves, stems, and roots.

Glyphosate is a white odorless solid that has a water solubility of 12 g/l and is not expected to bioconcentrate in aquatic biota based on water solubility. Glyphosate is strongly adsorbed to sediment colloids, silt, and suspended solids within the water column (Westerdahl and Getsinger 1988). Glyphosate is inactivated when sorbed to sediments. Because glyphosate is an acid, ionic rather than hydrophobic interactions are expected to account for its strong adsorption potential.

Glyphosate does not contain photolyzable or hydrolyzable groups and is not expected to degrade by either route (WSSA 1983). Biodegradation is considered the major fate process affecting glyphosate

persistence in aquatic environments. Glyphosate is biodegraded both aerobically and anaerobically by microorganisms present in soil, water and sediment.

In aquatic use, glyphosate has a minimum half-life of 2 weeks. Longer half-lives (7 to 10 weeks) have been observed in non-flowing natural water systems. QSAR estimates for aqueous biodegradation half-lives range from 2 to 15 days (Hunter et al. 1984). Glyphosate has no restriction on use of treated water for irrigation, recreation, or domestic purposes. The label prohibits application within 0.5 mile of potable water intakes. Glyphosate should not be used to retreat an area within 24 hours. Visible effects on most annual plants occur within 2 to 4 days, but perennial plants may not show effects for 7 days or more. In some cases, effects may not appear for up to 4 weeks depending on the physiological state of the plants. Visible effects are gradual wilting and yellowing of the plant, advancing to total browning and deterioration.

Toxicity Rodeo® has been found to have very low level of toxicity in mammals, birds, and fish, and only adversely impact fish and wildlife if valuable habitat were removed. Unless identified and avoided, some wetland and "unique" species may be killed if allowed to come into contact with Rodeo®. Removal of native species may allow infestation by noxious species such as purple loosestrife.

3. Environmental and Human Health Impacts

Earth

Soils Rodeo® is essentially unavailable for root uptake by plants since the herbicide binds tightly to soil particles upon contact (Monsanto 1988). Less than one percent of glyphosate in soil is absorbed through roots (USDA 1984a), although greater absorption by roots is possible under proper soil conditions (Ghassemi et al. 1981). Rodeo® is rapidly and completely biodegraded in soil by microorganisms (Rueppel et al. 1977). Rodeo® must be applied to emerged plant foliage for herbicidal activity to occur. Once absorbed, glyphosate is rapidly translocated throughout the plant.

Glyphosate is biodegraded both aerobically and anaerobically by microorganisms present in soil, water, and sediment. Average soil half-life is 60 days (WSSA 1983, Brandt 1984), and 90 percent of applied glyphosate is degraded within 6 months after treatment. Glyphosate applied to two Finnish agricultural fields persisted 69-127 days (Muller et al. 1981).

Glyphosate herbicide residues and metabolites were evaluated in forest brush field ecosystems in the Oregon Coast range aerially treated with 3 lb/acre glyphosate (Newton et al. 1984). The half-life ranged from 10 to 27 days in foliage and litter and 20 to 54 days in soil. The aminomethylphosphonic acid metabolite concentration averaged 0.4 percent of initial glyphosate levels in the first 24 hours, then rapidly declined over the following fourteen days. N-nitrosoglyphosate was nondetectable.

Adsorption to soil is believed to be through a phosphonic acid moiety, since the phosphate level in soil greatly influences the amount of glyphosate adsorbed (Sprankel et al. 1975a). Glyphosate adsorption is greater in soils saturated with aluminum and iron rather than with sodium and calcium (Sprankel et al. 1975b). At lower application rates, pH does not affect glyphosate binding to soil, while at high application rates, glyphosate binding decreases with increasing pH (Sprankel et al. 1975a).

Contamination of streams from surface spraying may be limited because the strong adsorption of glyphosate to soil reduces its mobility through leaching and surface runoff. Glyphosate had limited potential to leach in actual field studies and laboratory soil columns.

Sediments Glyphosate is inactivated (no measurable phytotoxic activity) when sorbed to sediments (Westerdahl and Getsinger 1988), and is strongly adsorbed to sediment colloids, silt, and suspended solids within the water column. Glyphosate is an acid; thus ionic rather than hydrophobic interactions account for its strong adsorption potential. Glyphosate does not contain photolyzable or hydrolyzable groups and is not expected to degrade by either route (WSSA 1983). In aquatic environments, glyphosate is degraded primarily by microorganisms, although at a slower rate than in soil (Ghassemi et al. 1981).

In aquatic situations, glyphosate binds to soil sediment and rapidly biodegrades with a half-life of 2 weeks. Water temperature, pH, water movement, and type of soil present can affect half-life values of glyphosate in aquatic sediments.

Air

Drift or overspray may be possible with the aerial application of glyphosate. However, the active ingredient in Rodeo® has negligible volatility, which minimizes the likelihood of aerial drift to non-target areas. Furthermore, restrictions on application procedures under windy conditions are included in label application instructions and in conditions to short-term modifications to water quality standards (or "permits") issued by Ecology before glyphosate is allowed to be applied. These instructions are expected to prevent aerial drift. Glyphosate is also applied with subsurface applicator devices; no aerial drift or overspray is expected with this application method.

Water

Surface Water Glyphosate is completely miscible; it mixes readily with water (Monsanto 1988). It has a water solubility of 12 g/l (WSSA 1983). There are no label restrictions against the use of water treated with glyphosate for irrigation, recreation, or domestic purposes. When applied according to label directions, residue levels in water will not exceed the acceptable level for the active ingredient (0.5 ppm) established by EPA. However, glyphosate cannot be applied within 1/2 mile of a domestic water intake. Crops irrigated with treated water will not be affected nor will they have unacceptable residue levels.

Glyphosate has a minimum half-life of 2 weeks in aquatic use, although longer half-lives (7 to 10 weeks) have been observed in non-flowing natural water systems. QSAR estimates for aqueous biodegradation half-lives range from 2 to 15 days (Hunter et al. 1984). Glyphosate is considered to dissipate rapidly from natural waters (Bronstad and Friestad 1985). Three pathways of dissipation are possible:

- 1) Photolytic,
- 2) AMPA, and
- 3) Adsorption to sediment, giving bound residues which slowly break down microbially under anaerobic conditions.

Glyphosate and its degradate AMPA are stable to hydrolysis in sterile, buffered water at pH 3, 6, and 9. In three natural waters (pH 4.2, 6.2, and 7.2), glyphosate degraded with half-lives of <50, 63, and >35 days, respectively. Addition of sediment to the three natural water systems increased the rate of dissipation of glyphosate from water via sorption to sediment. Glyphosate dissipated in pondwater with a half-life of between 14 and 21 days. In two canal waters, glyphosate was not detected 6 months post-treatment (EPA 1986).

Adverse impacts to surface water quality may occur after glyphosate treatment of aquatic vegetation. Large amounts of rapidly decaying vegetation in nonflowing waterbodies can result in oxygen depletion.

Oxygen depletion can lead to fish kills and the growth of microorganisms harmful to waterfowl (Monsanto 1988). The manufacturer suggests treating dense vegetation beds in alternate strips to avoid the possibility of oxygen depletion, and to allow 30 days between applications. Generally, Ecology will require the applicator to leave 25 percent of a waterbody's vegetation untreated to protect fish and waterfowl habitat.

Increased concentrations of organic and inorganic phosphorus may occur in the water column during plant decomposition following glyphosate treatment. Because phosphorus is often a limiting nutrient for algae, glyphosate treatment may result in rapid phytoplankton growth.

Drift of glyphosate into non-treatment areas may occur depending on wind conditions at the time of treatment. Restrictions on application procedures under windy conditions are included in label application instructions and in Ecology's "permit" for herbicide applications. These restrictions are expected to prevent drift.

Ground Water Rodeo® is strongly adsorbed by soil colloids, bottom silt, and suspended soil particles in water. In tests where soil columns were leached continuously with water for a period of 45 days, the active ingredient was not released from the soil (Monsanto 1988). Researchers concluded that the herbicide does not leach through the soil or move laterally into non-target areas.

Public Water Supplies In water, glyphosate binds to soil sediment and rapidly biodegrades with a minimum half-life of 2 weeks, and longer half-lives of up to 7 to 10 weeks. Water temperature, water movement, pH, and type of soil all affect half-life values. Somewhat longer half-life values for glyphosate have been observed in non-flowing natural water systems such as sphagnum bog (pH 4.23), 7 weeks; cattail swamp (pH 6.25), 9 weeks; and pond water (pH 7.33), 10 weeks (Monsanto 1988). Biodegradation by aerobic and anaerobic microorganisms is the major fate process affecting glyphosate persistence in aquatic environments. Glyphosate is inactivated when sorbed to sediments (Westerdahl and Getsinger 1988).

The label recommends against using Rodeo® within 0.5 miles of a potable water intake. Glyphosate becomes inactive when absorbed to sediments, thus, in-sediment impacts are expected to be minimal. Glyphosate has no restriction on use of treated water for irrigation, recreation, or domestic purposes. There are no swimming restrictions after treatment with Rodeo®. However, the label prohibits application of Rodeo® within 0.5 mile upstream of potable water intakes and states that Rodeo® must not be used to retreat an area within 24 hours.

Glyphosate is nonvolatile and has a negligible vapor pressure (WSSA 1983, Brandt 1984, Hunter et al. 1984). Thus, danger from inhalation is minimal. The tendency of glyphosate to transfer from water to the atmosphere is also negligible (Westerdahl and Getsinger 1988).

Chronic toxicity studies were conducted to determine the effects of prolonged, high level glyphosate exposure. Doses as high as 300 ppm incorporated into feed were provided daily for 2 years to rats, 1.5 years to mice, and two years to dogs. None of the test animals showed any evidence of cancer or other adverse effects. Monsanto (1988) concluded that Rodeo® does not cause cancer, mutations, nerve damage, birth defects, or adverse reproductive effects. Information on ingestion of Rodeo® was not found. However, ingestion of the Roundup® formulation of glyphosate has been reported to produce gastrointestinal discomfort, nausea, vomiting, and diarrhea. Roundup® is also reported as a mild skin irritant that can cause eye irritation.

Plants

Habitat The metabolism of glyphosate occurs via N-methylation and yields N-methylated glycines and phosphonic acids (EPA 1986). Degradation of ^{14}C glyphosate to $^{14}\text{CO}_2$ and the subsequent photofixation of respired $^{14}\text{CO}_2$ was demonstrated by the presence of ^{14}C -residues in control plants housed in close proximity to treated plants. The parent compound and its metabolite AMPA (aminomethylphosphonic acid) are the residues of concern in plants (EPA 1986).

Rodeo® has herbicidal activity on many annual and perennial grasses, broadleaf weeds, sedges, rushes, and woody plants. It is used to manage emerged and floating plants as well as ditchbank or shoreline aquatic weeds. Rodeo® is not effective on completely submerged plants, or those with most of their foliage under water (Monsanto 1988).

Annual grass and broadleaf weeds are best controlled at early growth stages after maximum weed emergence from the soil. Most perennial weeds are best controlled when treated at later growth stages when weeds are approaching maturity (Monsanto 1988). The Rodeo® label lists more than 90 plant species controlled by the herbicide. In the Pacific Northwest, Rodeo® is used primarily for control of waterlilies and purple loosestrife (T. McNabb, Aquatics Unlimited, pers. comm.).

Ecology does not support removal on non-noxious emergent (wetland) species except in controlled situations where the intent is to improve low quality wetlands (Category IV) or situations where wetlands have been created for other specific uses such as stormwater control, though ecology does not expect that chemicals will be used to control vegetation in stormwater facilities.

Noxious wetland species such as purple loosestrife are often invasive and out-compete native plant species. Invasion of wetlands by purple loosestrife is predicted to cost the state over \$500,000 in management costs annually. Treatment of invasive species with glyphosate may allow native species to return and may result in a greater diversity of species rather than a monoculture. If eradication is chosen as an objective, the plan should include long-term measures (including mitigation) to ensure no net loss of wetlands.

Additionally, glyphosate treatment can affect native species. Use of glyphosate may result in positive or negative impacts depending on vegetation in the specific waterbody. Negative impacts could include invasion by less desirable species such as reed canarygrass or soft rush. Another potential negative impact of glyphosate treatment would be destruction of perimeter or riparian emergent vegetation. Loss of perimeter vegetation may increase shoreline erosion and decrease the treated waterbody's value as wildlife habitat.

Animals

Glyphosate and its formulation as Rodeo® have a very low level of toxicity in mammals, birds, and fish (Monsanto 1988). Acute toxicological values for invertebrates using various formulations of glyphosate include the following:

Daphnia magna	48-hr LC ₅₀	930 mg/l	Rodeo
	48-hr LC ₅₀	780 mg/l	Glyphosate (tech. grade)
Honeybee	48-hr LC ₅₀	>100 ug/bee	Glyphosate (tech. grade)
Shrimp	96-hr LC ₅₀	281 mg/l	Glyphosate (tech. grade)
Fiddler crab	96-hr LC ₅₀	934 mg/l	Glyphosate (tech. grade)

Midge larvae	48-hr LC ₅₀	>10 mg/l	Glyphosate (tech. grade)
(<u>Chironomus plumosus</u>)	48-hr LC ₅₀	55 mg/l	Glyphosate (tech. grade)

With the exception of one value for fiddler crab (>10 mg/l = slightly toxic), all values were determined to be practically nontoxic. In actual application of Rodeo®, it is unlikely that glyphosate concentrations would ever approach toxic levels because of its binding capacity to soil particles and its rapid microbial degradation (Monsanto 1988).

Acute toxicological values for fish using various formulations of glyphosate follow:

Carp	96-hr LC ₅₀	>10,000 mg/l	Rodeo
	115 mg/l		Glyphosate (tech. grade)
Rainbow trout	24-hr LC ₅₀	140 mg/l	Glyphosate (tech. grade)
	96-hr LC ₅₀	>1,000 mg/l	Rodeo
	96-hr LC ₅₀	86 mg/l	Glyphosate (tech. grade)
	96-hr LC ₅₀	140 mg/l	Glyphosate (tech. grade)
Bluegill	96-hr LC ₅₀	>1,000 mg/l	Rodeo
sunfish	96-hr LC ₅₀	120 mg/l	Glyphosate (tech. grade)
	24-hr LC ₅₀	150 mg/l	Glyphosate (tech. grade)
	96-hr LC ₅₀	140 mg/l	Glyphosate (tech. grade)
Harlequin fish	96-hr LC ₅₀	>1,000 mg/l	Rodeo
	96-hr LC ₅₀	168 mg/l	Glyphosate (tech. grade)
Channel	24-hr LC ₅₀	130 mg/l	Glyphosate (tech. grade)
catfish	96-hr LC ₅₀	130 mg/l	Glyphosate (tech. grade)
Fathead	24-hr LC ₅₀	97 mg/l	Glyphosate (tech. grade)
minnows	96-hr LC ₅₀	97 mg/l	Glyphosate (tech. grade).

Again all values (except trout, 86 mg/l = slightly toxic, technical grade glyphosate) are considered practically nontoxic (Monsanto 1988, EPA 1986).

Additional information is available on acute toxicity of the terrestrial formulation of glyphosate (Roundup®) but was not included in this summary. In general, the toxicity of Roundup® to aquatic organisms is greater than of Rodeo®, due primarily to the surfactant POEA (polyethoxylated tallow amine) included in the Roundup® formulation.

Acute toxicity of technical grade glyphosate was tested on ducks and quail. Five-day LC₅₀ values were >4,640 mg/l for each, or practically nontoxic (Monsanto 1988, EPA 1986). The acceptable level for the active ingredient established by EPA is 0.5 ppm. Chronic studies were conducted on rodents and dogs to determine effects of prolonged high level glyphosate exposure. Doses as high as 300 ppm incorporated into feed were fed to rats for 2 years, mice for 1.5 years, and to dogs for 2 years. No test animal showed any evidence of cancer or other adverse effects. Information regarding oncogenicity is equivocal.

Threatened and Endangered Species Treatment with herbicides has the potential to affect submersed or emersed plant species listed by the federal government as rare, threatened, or endangered. These species may be aquatic or may occur along the banks of waterways. Applications for short-term modifications to water quality standards for each specific site should include a review of the rare, threatened, or endangered species listed by US Fish and Wildlife, and of "proposed sensitive" plants and animals listed by Washington State National Heritage Data System.

The bioconcentration factor (BCF) for glyphosate is low (Westerdahl and Getsinger 1988). Based on water solubility, glyphosate is not expected to bioconcentrate in aquatic biota. Brandt (1984) used glyphosate concentrations 3 to 4 times the recommended levels in laboratory studies; BCF values in experimental fish tissue ranged from 0.2 to 0.3 after a 10- to 14-day exposure period. A BCF value of 100 is usually regarded as a significant factor.

Further studies show minimal tissue retention and rapid elimination of the active ingredient from several animal species including mammals, birds, and fish (Monsanto 1988). Thus, there is a very low probability that glyphosate will bioaccumulate or biomagnify in animals. Since the BCF for glyphosate is low, the need for concern for bald eagles and other threatened or endangered predators of fish in treated areas is also low.

Water, Land and Shoreline Use

Swimming There are no swimming restrictions associated with glyphosate treatment. Treatment is expected to improve swimming conditions in areas of dense aquatic vegetation. Increased open water areas would improve recreational activities such as boating.

Fishing Glyphosate has no fishing restrictions associated with its use, and fish are not affected at treatment concentrations. Navigation to and from fishing areas may be improved after treatment with glyphosate. Decreased lily densities may reduce habitat for recreational fish species. It also often allows submerged species to colonize the area, providing good habitat. In addition, decreased populations of purple loosestrife following glyphosate treatment may improve habitat for wildlife and waterfowl by allowing increased species diversity of native plants.

Agriculture Glyphosate, in the Roundup® formulation for terrestrial application, is used extensively to control weeds in agricultural crops prior to their emergence and in spot treatment. These crops include alfalfa, artichoke, barley, beans, beet greens, beets, broccoli, cabbage, carrot, cauliflower, celery, chicory, corn, cotton, forage grasses, forage legumes, horseradish, kale, lentils, lettuce, mustard greens, oats, okra, onion, parsnips, peanuts, peas, potatoes, radish, rice, rutabaga, sorghum, soybeans, spinach, and wheat. Rodeo is formulated for aquatic use and is not used for crop treatment.

With either formulation, spray or drift of glyphosate outside the target area is expected to kill crops with foliage, green stems, or fruit (Monsanto 1988). However, glyphosate has no restriction on use of treated water for irrigation. Glyphosate is highly immobile in soil. The rapid disappearance of herbicidal activity when applied to soil is due in part to inactivation through adsorption. Volatilization does not occur and leaching is practically negligible. Disappearance through degradation is often slow. When applied to plant roots, glyphosate has a low intrinsic toxicity. Torstensson concludes in his 1985 review of glyphosate in soils that the herbicide will not cause any unexpected damage after application to soil.

Additional Information Some early health damage tests submitted to EPA for glyphosate registration were performed by Industrial Bio-Test Laboratory and were found in 1976 to be invalid. Most of the tests were repeated by 1984 and were reviewed by EPA (NCAP 1986). EPA (1987) reports that it classified glyphosate in Group D: not classified. This category is for substances with inadequate animal evidence of carcinogenicity. The Science Advisory Board (Pesticides) considered the evidence of carcinogenicity in animals equivocal and the Office of Pesticide Programs has requested the manufacturer to conduct another study in animals (EPA 1987).

Glyphosate may include N-nitrosoglyphosate (NNG) as a trace contaminant, or the compound may be formed in the environment when combined with nitrite (NCAP 1986). This would have potential importance in terrestrial applications of Roundup® in soils treated with nitrogen fertilizers. Many N-nitroso compounds are potent animal carcinogens (Khan and Young 1977).

Navigation Waterlilies can form floating mats after glyphosate treatments and these can block navigation.

4. Mitigation

The review of glyphosate does not reveal a need for conditions over and above those presented in the discussion of general mitigation measures. General mitigation measures include provisions that require adherence to all label restrictions, which would include the restriction against using glyphosate within 1/2 mile of a potable water intake.

Environmental impacts associated with the use of Rodeo® are expected to be minimal. Research indicates that the only identified adverse impacts to water quality associated with glyphosate applied at the label rate are those caused by large areas of decaying plant material, which can result in oxygen depletion that may cause fish kills and growth of micro-organisms harmful to waterfowl. Limiting the amount or area of plants to be treated could mitigate these impacts.

Notification Use of any aquatic herbicide requires notification of affected shoreline owners. Current requirements for glyphosate include notification of lake residents within 1/2 mile of the treatment area. Before herbicide treatment, the applicator must post the shoreline with signs stating the herbicide used and restrictions associated with that herbicide. The applicator may be required to mark treatment area boundaries on the water with buoys. In some cases, newspaper notices may be required by Ecology.

Spills Liquid spills on impervious surfaces should be contained or diked, and absorbed with absorbent clays. Contaminated absorbent should be placed in a plastic-lined container and disposed. The spill surface should be sorbed with an industrial type detergent and rinsed. Liquid spills that soak into the ground should be dug-up, placed in plastic-lined metal drums, and disposed.

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