Lake Iroquois – application #2240-ANC
Technical updates to VT DEC in response to public comments

1. Herbicide concentrations sampling locations.
    - See attached map.

2. Updated Lake Iroquois Integrated Management Plan.
    - See attached document.

3. Technical description on Sonar breakdown.
    - Please refer to the following documents, which are attached
        - Appendix III of the Massachusetts Final Generic Environmental Impact Report – pages 91-100
        - Wisconsin Department of Natural Resources – Fluridone Chemical Fact Sheet
        - Use of the Registered Aquatic Herbicide Fluridone (Sonar) in the State of New York – Generic Environmental Impact Statement
    - Additionally, we understand that this information has previously been given to the State of Vermont in a Confidential Statement of Formulation by SePRO, the manufacturer of Sonar

4. Technical resource identifying fluridone ecotoxicity.
    - Please refer to the following documents, which are attached
        - Appendix III of the Massachusetts Final Generic Environmental Impact Report – pages 91-100
        - Supplemental Environmental Impact Statement Assessment of Aquatic Herbicides, Washington State Department of Ecology – pages 115-128
        - Fluridone Ecological Risk Assessment, Final Report – ENSR International
        - Use of the Registered Aquatic Herbicide Fluridone (Sonar) in the State of New York – Generic Environmental Impact Statement
        - Technical Perspectives on Use of Sonar Pellet Formulations and Potential Risks to Threatened Mussels, Mark Heilman, Ph.D, SePRO (although this is about the pellet formulation, we have included it as an overall impact to mollusks from fluridone)
        - Permit #2001-C08 RE: Use of Sonar A.S. to control Eurasian watermilfoil in Lake St. Catherine, VT DEC ANC Permit Program – Section 4A & 4B, pages 17-22
    - Additionally, we understand that this information has previously been given to the State of Vermont in a Confidential Statement of Formulation by SePRO, the manufacturer of Sonar

5. Narrative regarding objective/goal #1.
    - See aforementioned attached Integrated Management Plan document.

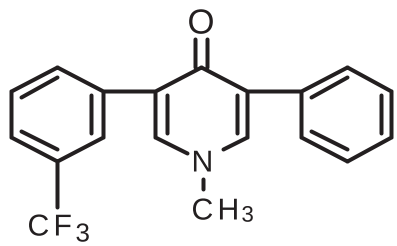
6. Web address for updated public notification.
    - http://www.lakeiroquois.org/lake-status

7. Public notification plan.
    - To be supplied by Jamie Carroll/LIA at a later date.

8. Notice of Addition of Permittee – identifying LIA as a decision-maker
    - See attached document.

9. Pesticide General Permit application.
    - See attached documents.

III.5 FLURIDONE



FLURIDONE

SUMMARY

Fluridone (1-methyl-3-phenyl-5-[3-(trifluoromethyl)phenyl]-4(1H)-pyridinone) is a selective systemic aquatic herbicide used to control primarily broad-leaved, submerged aquatic macrophyte species including Eurasian watermilfoil, curly-leaf pondweed as well as native pondweeds (McLaren/Hart, 1995). It is used to treat ponds, lakes, reservoirs, canals and rivers. Fluridone is stable to oxidation and hydrolysis (McCowen *et al.*, 1979 as cited in Aquatic Plant Identification and Herbicide Use Guide, 1988). Volatilization of fluridone is insignificant (Muir and Grift, 1982 as cited in Aquatic Plant Identification Guide, 1988). Breakdown of fluridone in the aquatic environment occurs mostly through photolysis. Other fate processes include plant uptake and adsorption to soil and suspended colloids (Joyce and Ramey, 1986). Some microbial degradation of fluridone has also been reported (Muir and Grift, 1982 as cited in McLaren/Hart, 1995). Fluridone is taken up in fish but studies demonstrate that fish tissue concentrations generally reflect water concentrations and that fish concentrations rapidly clear when fluridone residues are removed from the water (West *et al.*, 1983 and Muir and Grift, 1982 as cited in McLaren/Hart, 1995). There are no restrictions on the use of fluridone to treat water used for swimming or domestic purposes. Care should be taken when applying Fluridone within one-fourth mile of any potable water intake (WSDOE, 1992).

The U.S. Environmental Protection Agency (USEPA) approved the label for Sonar on March 31, 1986 (McLaren/Hart, 1995).

REGISTERED PRODUCTS IN MASSACHUSETTS

The current list of aquatic herbicides containing fluridone that are registered in Massachusetts can be accessed at http://www.state.ma.us/dfa/pesticides/water/Aquatic/Herbicides.htm on the Massachusetts Department of Agricultural Resources (DAR) Aquatic Pesticide Website. The DAR updates this list regularly with changes. In addition, the DAR can be contacted directly at (617) 626-1700 for more specific questions regarding these products.

FLURIDONE USES AND APPLICATION

Fluridone is used to manage aquatic vegetation in fresh water ponds, lakes, reservoirs, canals and rivers (Cockreham, pers. comm., 1996). It is absorbed from the water by the shoots of submerged plants and from the hydrosoil by the roots of aquatic vascular plants. The effectiveness of fluridone depends on the degree to which the herbicide maintains contact with plants. Rapid water movement or any dilution of this herbicide in water will reduce its effectiveness (Dow Elanco, 1992; Aquatic Plant Identification and Herbicide Use Guide, 1988; WSDOE, 1992).

Application of fluridone may be made in several ways depending on the formulation used. The liquid suspension may be applied as a spray to the water surface, subsurface or along the bottom of the water body using specialized equipment. The pellet can be spread on the water surface (WSSA, 1983). Water should be used as a carrier during application of the liquid fluridone suspension. No surfactant is specified for use during application.

When treating ponds, application should be made to the entire water body. When treating lakes and reservoirs, plots no smaller than ten surface acres should be treated. In addition, areas with a large linear aspect (such as boat lanes and narrow shorelines) should not be treated (Aquatic Plant Identification and Herbicide Use Guide, 1988).

Application of fluridone may be made prior to active growth of aquatic weeds or any time during the spring or summer when weeds are visible (WSSA, 1983; Aquatic Plant Identification and Herbicide Use Guide, 1988).

Caution should be used when applying fluridone within one-fourth mile of any functioning potable water intake.

The plant selectivity of fluridone is dependent upon dose, application timing and formulation used. For specific information on recommended application rates for a particular product, the product label should be consulted. The USEPA Office of Pesticide Programs (OPP) has a link to a database of product pesticide labels at http://www.epa.gov/pesticides/pestlabels/. A list of the weeds that these products control, which has been compiled from the Environmental Protection Agency (EPA) registration labels for these products, is contained in Table III.5-1.

MECHANISM OF ACTION

Fluridone produces its toxic effect in plants by inhibiting synthesis of carotenes (pigments that protect chlorophyll molecules from photodegradation). The absence of carotenes causes degradation or "bleaching" of chlorophyll by sunlight from plants. Plants become whitish-pink or chlorotic at growing points and die slowly. This slow dying-off of plants (i.e., 30-90 days) (Cockreham, pers. comm., 1996) reduces the instantaneous oxygen demand caused by plants dying off and decomposing all at once (Joyce and Ramey, 1986). The herbicidal effects of fluridone usually appear within 7-10 days. Species susceptibility to fluridone may vary depending on time of year, stage of growth and water movement (McLaren/Hart, 1995).

Table III.5-1. List of Aquatic Plants Controlled by Fluridone

Common Name	Scientific Name
American Lotus	*Nelumbo lutea*
Bladderwort	*Utricularia* spp.
Common Coontail	*Ceratophyllum demersum*
Common Duckweed	*Lemna minor*
Common Elodea	*Elodea canadensis*
Egeria, Brazilian Elodea	*Egeria densa*
Fanwort	*Cabomba caroliniana*
Hydrilla	*Hydrilla verticillata*
Naiad	*Najas* spp.
Pondweed (except Illinois)	*Potamogeton* spp.
Watermilfoil (including Eurasian Watermilfoil)	*Myriophyllum* spp. (including *M. spicatum)*
Spatterdock	*Nuphar* spp.
Waterlily	*Nymphaea* spp.
Waterprimrose (including Waterpurslane)	*Ludwigia* spp. (including *Ludwigia palustris*)
Watershield	*Brasenia schreberi*

(McLaren/Hart, 1995; SePRO, 1994)

ENVIRONMENTAL FATE/TRANSPORT

The major fate process affecting fluridone persistence in aqueous environments is photolysis. Thus any factors which affect sunlight intensity and/or penetration of light into the water column will affect the dissipation rate of fluridone (Joyce and Ramey, 1986). Other factors affecting dissipation include geographic location, date of application, water depth, turbidity, weather and weed cover (West *et al.*, 1983 as cited in McLaren/Hart, 1995). Microbial degradation is also reported to occur in laboratories, but photolysis generally occurs much more quickly (Muir and Grift, 1982 as cited in McLaren/Hart, 1995). Other secondary fate processes include adsorption to soil and suspended colloids and plant uptake (Joyce and Ramey, 1986).

Fluridone will adhere to sediment particles/organics in the sediment. Eventually, the fluridone will desorb and photodegrade into the water column from the hydrosoil (Elanco, 1981 as cited in McLaren/Hart, 1995). The pH of the water can affect this rate (with the lower the pH, the higher the adsorption rate (Malik and Drennan, 1990 as cited in McLaren/Hart, 1995).

Fluridone is taken up in fish tissue. Fluridone fish concentrations generally reflect the concentrations of fluridone in the water (McLaren/Hart, 1995). When fluridone residues are removed from the water column, the fluridone concentration from fish tissue clears (West *et al.*, 1983; Muir *et al.*, 1983 as cited in McLaren/Hart). Based on a low bioaccumulation rate in fish and high levels of fluridone necessary to produce toxic responses in mammals and birds, it is not expected that fish-eating animals would be affected by fluridone used at recommended (registered) application rates (McLaren/Hart, 1995).

The primary metabolite of fluridone degradation in fish was identified as 1-methyl-3-(4-hydroxyphenol)-5-[3-trifluoromethyl)phenyl]-4[1H]-pyridone (West *et al.*, 1983 as cited in McLaren/Hart, 1995). This compound was also identified as a minor metabolite in water and hydrosoil (Muir and Grift, 1982 as cited in McLaren/Hart, 1995). 1,4-dihydro-1-methyl-4-oxo-5-[3-(trifluoromethyl)phenyl]-3-pyridone was also identified as the major hydrosoil metabolite in hydrosoil studies conducted in the laboratory; however, this compound has not been identified in the hydrosoil of small ponds under natural conditions (West *et al.*, 1983 as cited in McLaren/Hart, 1995). A number of other metabolites including benzaldehyde, 3-(trifluoromethyl)-benzaldehyde, benzoic acid and 3-(trifluoromethyl)-benzoic acid were produced as photolytic breakdown products in one laboratory study (Saunders and Mosier, 1983, as cited in McLaren/Hart, 1995). N-methylformamide (NMF) was produced in another study. However, NMF has not been identified as a breakdown product under natural conditions (Saunders and Mosier, 1983 as cited in McLaren/Hart, 1995).

The half-life of fluridone in water of small, artificial ponds ranged from 1-7 days. In hydrosoils, the compound persisted for 8 weeks to one year (Joyce and Ramey, 1986; WSDOE, 1992). Fluridone has a water solubility of 12 mg/l and an octanol-water partition coefficient (K_{ow}) of 74.1 (Elanco Products Company, 1985 as cited in Aquatic Plant Identification and Herbicide Use Guide, 1988). Fluridone is stable to oxidation and hydrolysis (McCowen *et al.*, 1979). Volatilization of fluridone is not expected to be a significant process, (Muir and Grift, 1982 as cited in Aquatic Plant Identification and Herbicide Use Guide, 1988).

PHARMACOKINETICS

Metabolism and distribution studies have shown that fluridone is absorbed and excreted in the feces within 72 hours of oral administration to rats (McLaren/Hart, 1995). No bioaccumulation of fluridone was noted. 90% of the absorbed fluridone was cleared in 96 hours (USEPA, 1988).

HEALTH EFFECTS

Avian:

Fluridone has very low toxicity to birds. A number of acute toxicity studies were conducted in various bird species. An oral LD50 value of >2,000 mg/kg was obtained for bobwhite quail. The EPA considers this value to represent slight toxicity (USEPA, 1986). An LD50 of >2,000 was identified for mallard ducks (WSDOE, 1992). Oral LC50 values of > 5,000 ppm were identified for bobwhite quail and mallard duck (USEPA, 1986). No impairment on reproduction for the above species was noted up to a dietary exposure concentration of 1,000 ppm (USEPA, 1986). In other studies, an LC50 value of about 10,000 ppm was identified for bobwhite quail and an LC50 value of >20,000 ppm was identified for mallard duck (WSDOE, 1992).

Mammalian:

Acute:

Most of the available information on the toxic effects of fluridone comes from studies conducted by the industry on various formulations of the product. Generally, the acute toxicity of fluridone is low. The LD_{50} for both rats and mice exposed through ingestion to technical grade fluridone is greater than 10,000 mg/kg. The oral LD_{50}s for cats and dogs exposed to technical grade fluridone are 250 mg/kg and 500 mg/kg, respectively. The LD_{50} for rabbits exposed through the skin to technical grade fluridone is greater than 2,000 mg/kg (Elanco, 1981 as cited in McLaren/Hart, 1995).

Fluridone was found to produce eye irritation in rabbits with effects including redness, corneal dullness and conjunctivitis. Fluridone was found to be neither irritating nor a sensitizer to rabbit skin at 2,000 mg/kg (USEPA, 1988).

Subchronic/Chronic:

In a three-week study in which fluridone was applied to rabbit skin daily at doses ranging from 192-768 mg/kg/day, dose-dependent skin irritation was produced at all doses. No systemic effects were noted at any dose. An increase in organ weight was noted at 384 mg/kg/day (USEPA, 1988).

In a three-month subchronic feeding study with fluridone, no treatment-related effects were noted in rats administered doses of 62 mg/kg or in mice administered doses of 330 mg/kg (Elanco, 1981 as cited in McLaren/Hart, 1995). A dietary level of fluridone of 16.5 mg/kg/day administered to mice for three months resulted in a partial enlargement of livers. A dietary level of 166 mg/kg administered to rats for three months resulted in an increase in liver weights. A No Observed Effect Level (NOEL) of 30 mg/kg/day was identified in rats administered fluridone in the diet for three months (USEPA, 1988). A concentration of 0.033% of fluridone fed to mice for three months produced morphologic changes in the liver and an increase in absolute liver weights in male mice (USEPA, 1988). In a study conducted with dogs, daily dietary fluridone levels up to 200 mg/kg/day did not result in any treatment-related effects (Elanco, 1978 as cited in USEPA, 1990).

In a one-year chronic study in which dogs were administered fluridone by capsule in food, a number of effects including weight loss, an increase in liver weight and an increase in liver enzymes were noted at a dose level of 150 mg/kg/day. A NOEL of 75 mg/kg/day was identified (USEPA, 1988). In a two-year feeding study in which mice were administered fluridone concentrations in the diet of up to 330 ppm fluridone, there was an increase in liver enzymes in males exposed at 330 ppm. No other toxic effects or lesions were noted at any of the doses (USEPA, 1988). In another two-year study, rats were exposed to doses of 0, 8, 25 and 81 mg/kg/day. At 25 mg/kg/day, rats experienced inflammation in the kidney, atrophy of the testes, inflammation of the cornea, weight loss and decreased organ weights (USEPA, 1988; USEPA, 1990).

Developmental/Reproductive:

In a study in which rats were exposed to up to 200 mg/kg/day of fluridone, neither maternal nor fetotoxic effects were noted (USEPA, 1988). In a three-generation study conducted in rats exposed to fluridone at a dose of 100 mg/kg/day, no teratogenic or maternal effects were noted. However, a dose of 100 mg/kg/day was found to be toxic to rat pups (USEPA, 1988; USEPA 1990). In a teratology study in which rabbits were exposed to fluridone doses of up to 750 mg/kg/day, a level of 300 mg/kg resulted in maternal effects including a decrease in body weight gain and abortion. Fetal effects, also noted at this level, included resorptions (USEPA, 1988). No teratogenic effects were noted (USEPA, 1990). In a pilot study in which rabbits were exposed to fluridone at doses of 0, 250, 500, 750 and 1,000 ppm, a maternal NOEL of 500 mg/kg was identified. A level of 750 mg/kg produced a maternal loss in body weight. A NOEL of 250 mg/kg/day was identified for fetal effects. At 500 mg/kg/day, fetal resorptions occurred (USEPA 1988). In another study, rats were administered doses by oral gavage of 0, 100, 300 and 1,000 mg/kg/day. A maternal NOEL of 100 mg/kg/day was identified. At 300 mg/kg/day, there was a decrease in maternal body weight. The NOEL for developmental effects was identified as 300 mg/kg/day. At 1,000 mg/kg/day, fetal effects included a decrease in fetal weight and delayed ossification. The NOEL for teratogenic effects was greater than 1,000 mg/kg/day (USEPA 1988).

Mutagenicity:

Fluridone was not found to be mutagenic in several test assays. Fluridone produced negative results in the Ames assay and did not induce sister chromatid exchange in Chinese hamster bone marrow cells. In addition, fluridone did not promote unscheduled DNA synthesis in rat hepatocytes (USEPA, 1988).

Carcinogenicity:

Based on negative cancer findings in the two chronic toxicity studies discussed above, there is no evidence indicating that fluridone is carcinogenic. The EPA Health Effects Division has designated fluridone as a Group E carcinogen (i.e., having evidence of noncarcinogenicity for humans) by the old EPA classification system. Under the new cancer classification system (USEPA, 1995), an E classification would correspond to a weight-of-evidence descriptor of “not likely to be carcinogenic to humans”..

Available Toxicity Criteria:

The EPA Carcinogen Risk Assessment Verification Endeavor (CRAVE) (RfD/RfC) workgroup has developed an oral Reference Dose (RfD) of 0.08 mg/kg/day for fluridone based on one of the two-year rat feeding studies conducted by Elanco cited earlier (USEPA, 1990). The EPA Office of Pesticide Programs (OPP) has calculated the same RfD value based on the same study (USEPA, 1995). The RfD is an estimate, (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime (USEPA, 1990).

The EPA has designated an acceptable residue level for fluridone in potable water of 0.15 ppm. This level is based on the maximum application rate for fluridone as registered under FIFRA (Federal Insecticide, Fungicide and Rodenticide Act) (USEPA, 1986 as cited in McLaren/Hart, 1995). The EPA has also established a tolerance of 0.5 ppm for residues of fluridone and its primary metabolites in fish and crayfish. In addition, EPA has established tolerances for crops irrigated with water containing fluridone residue concentrations at 0.15 ppm as well as for a number of raw, agricultural commodities (USEPA, 1986 as cited in McLaren/Hart, 1995).

ECOLOGICAL TOXICITY

Aquatic Organisms:

A number of studies have been conducted with fluridone to determine the LD50 or LC50 values for a variety of organisms. The LD50 (or LC50) is the dose (or concentration) to which a particular species is exposed, which results in the death of 50% of the test population. The EPA has cited the results of a number of these studies. EPA considers these studies to demonstrate moderate toxicity. These studies are listed in the Table III 5-2.

In addition, a Maximum acceptable theoretical concentration (MATC) value for fathead minnow (second generation fry) was calculated to be between 0.48 mg/l and 0.96 mg/l, meaning no treatment-related effects were noted at or below 0.48 mg/l. Total length of 3-day old fry was reduced at 2 mg/l fluridone (USEPA, 1986).

No adverse effects were noted on crayfish, bass, bluegill, catfish, long-neck soft-shelled turtles, frogs, water snakes and waterfowl from the use of 0.1 to 1.0 ppm fluridone during field experiments (Arnold, 1979, McCowen et al., 1979 as cited in WSDOE, 1992). Application of 1.0 ppm fluridone to zooplankton caused a reduction in population, but the population quickly recovered. Application of 0.3 ppm did not cause a change in the total number of benthic organisms whereas application of 1.0 ppm did cause a change (Parka *et al.*, 1978 as cited in WSDOE, 1992). An aqueous solution of fluridone caused a reduction in population of the amphipod *Hyalella azteca* when applied at a rate of 1.0 ppm but not when applied at a rate of 0.3 ppm (Arnold, 1979 as cited in McLaren/Hart, 1995). Fish abundance and community structure remained unchanged in ponds exposed to a fluridone concentration level of 0.125 ppm (Struve *et al*. 1991 as cited in McLaren/Hart, 1995). LC50 values for a variety of microscopic crustaceans including *Diaptomus*, sp., *Eucyclops* sp, *Alonella* sp., and *Cypria* sp., ranged from 8.0 - 13.0 ppm (Naqvi and Hawkins, 1989 as cited in McLaren/Hart, 1995).

Table III.5-2. Acute Toxicity Tests

SPECIES	TEST TYPE	VALUE
Daphnia magna	48-hr LC50	6.3 mg/l
Bluegill	96-hr LC50	12 mg/l
Rainbow trout	96-hr LC50	11.7 mg/l
Sheepshead minnow	96-hr LC50	10.91 mg/l
Oyster embryo larvae	48-hr LC50	16.51 mg/l

(USEPA, 1986)

One group of investigators conducted extensive acute toxicity tests on a variety of aquatic invertebrates including amphipods, midges, daphnids, crayfish, blue crabs, eastern oysters and pink shrimp. The average 48-hour or 96-hour LC50 or EC50 (concentration at which 50% of the organisms exhibit an effect) was calculated as 4.3 ± 3.7 ppm (Hamelink *et al.*, 1986 as cited in McLaren/Hart, 1995). The same investigators also conducted studies with a variety of fish including rainbow trout, fathead minnows, channel catfish, bluegills and sheepshead minnows. A 96-hour LC50 value of 10.4 ± 3.9 was calculated (Hamelink *et al.*, 1986 as cited in McLaren/Hart, 1995).

Daphnids, amphipods and midge larvae exposed chronically to fluridone concentrations of 0.2, 0.6 and 0.6 ppm as well as catfish fry exposed to fluridone concentrations of 0.5 ppm showed no treatment-related significant effects. Exposure to concentrations of 1 ppm produced a decreased growth rate of catfish fry and concentrations of 0.95 and 1.9 ppm produced a decreased survival rate of fathead minnows within 30 days after hatching (Hamelink *et al.*, 1986 as cited in McLaren/Hart, 1995).

Plants:

Fluridone selectively controls a number of broad-leaved submerged and floating aquatic macrophyte species as specified by its EPA label. In addition, the literature contains reports of fluridone's variable efficacy in controlling other species. The efficacy of fluridone is very dependent on contact time with plants. Thus, fluridone should be applied during periods of minimum water movement. Factors related to fluridone's variable efficacy include temperature, pH and light levels (Wells *et al*. 1986 as cited in WSDOE, 1992). In addition, one investigator found that in *Hydrilla* exposed 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 (Spencer and Ksander, 1989 as cited in WSDOE, 1992).

Fluridone did not appear to adversely affect desirable phytoplankton but some reduction in population of the less desirable species given as *Anabaena* and *Anacystis* occurred upon application of fluridone at levels of 0.3 and 0.1 ppm (Parka et al, 1978 as cited in WSDOE, 1992). A drastic reduction in phytoplankton population in Greek ponds including the disappearance within two months of a population of Cyanophyceae (Cyanobacteria) occurred after fluridone application. Diatom populations, a more desirable species, increased significantly, especially epiphytic and benthic species (Kamarianos *et al.*, 1989 as cited in WSDOE, 1992). No sufficient reduction in phytoplankton densities was noted when they were consistently exposed to a fluridone concentration of 0.125 ppm (Struve *et al.*, 1991 as cited in McLaren/Hart, 1995).

An aqueous solution of fluridone applied at a concentration of 1.0 ppm produced a significant reduction in a zooplankton population whereas a concentration of 0.3 ppm had no effect. The 1.0 ppm population returned to pretreatment levels within 43 days (Arnold, 1979 as cited in McLaren/Hart, 1995).

Table III.5-3. Properties of Fluridone

CAS #:	59756-60-4
Synonyms:	1-methyl-3-phenyl-5-[3-(trifluoromethyl)phenyl-4(1H)-pyridinone;
Molecular formula	$C_{19}H_{14}F_3NO$
Molecular weight	329.3
Physical properties	white, crystalline solid
Melting point	154-155^{o}C
Vapor pressure	< 1 x 10^{-7} mm Hg at 25^{o}C
Photolysis half-life	1-6 days
Hydrolysis half-life	stable
Biodegradation half-life	2-60 days (based on overall half-life)
K_{ow}	74.1 at 20^o C
K_{oc}	~350-2460 ml/g
BCF	0.9-15.5
Water solubility	12 mg/l at 25^o C and pH 7

(Reinert and Rodgers, 1987; WSSA, 1983; Aquatic Plant Identification and Herbicide UseGuide, 1988; WSSA, 1994)

Fluridone References

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January 2012

Fluridone Chemical Fact Sheet

Formulations

Fluridone is an aquatic herbicide that was initially registered with the EPA in 1986. The active ingredient is 1-methyl-3-phenyl-5-3-(trifluoromethyl)phenyl|-41H|-pyridinone. Both liquid and slow-release granular formulations are available. Fluridone is sold under the brand names Avast!, Sonar, and Whitecap (product names are provided solely for your reference and should not be considered endorsements).

Aquatic Use and Considerations

Fluridone is an herbicide that stops the plant from making a protective pigment that keeps chlorophyll from breaking down in the sun. Treated plants will turn white or pink at the growing tips after a week and will die in one to two months after treatment as it is unable to make food for itself. It is only effective if plants are growing at the time of treatment.

Fluridone is used at very low concentrations, but a very long contact time is required (45-90 days). If the fluridone is removed before the plants die, they will once again be able to produce chlorophyll and grow.

Fluridone moves rapidly through water, so it is usually applied as a whole-lake treatment to an entire waterbody or basin. There are pellet slow-release formulations that may be used as spot treatments, but the efficacy of this is undetermined. Fluridone has been applied to rivers through a drip system to maintain the concentration for the required contact time.

Plants vary in their susceptibility to fluridone, so typically some species will not be affected even though the entire waterbody is treated.

Plants have been shown to develop resistance to repeated fluridone use, so it is recommended to rotate herbicides with different modes of action when using fluridone as a control.

Fluridone is effective at treating the invasive Eurasian watermilfoil (*Myriophyllum spicatum*). It also is commonly used for control of invasive hydrilla (*Hydrilla verticillata*) and water hyacinth (*Eichhornia crassipes*), neither of which are present in Wisconsin yet. Desirable native species that are usually affected at concentrations used to treat the invasives include native milfoils, coontail (*Ceratophyllum demersum*), naiads (*Najas* spp.), elodea (*Elodea canadensis*) and duckweeds (*Lemna* spp.). Lilies (*Nymphaea* spp. and *Nuphar* spp.) and bladderworts (Utricularia spp.) also can be affected.

Post-Treatment Water Use Restrictions

There are no restrictions on swimming, eating fish from treated water bodies, human drinking water or pet/livestock drinking water. Depending on the type of waterbody treated and the type of plant being watered, irrigation restrictions may apply for up to 30 days. Certain plants, such as tomatoes and peppers and newly seeded lawn, should not be watered with treated water until the concentration is less than 5 parts per billion (ppb).

Herbicide Degradation, Persistence and Trace Contaminants

The half-life of fluridone (the time it takes for half of the active ingredient to degrade) ranges from 4 to 97 days depending on water conditions. After treatment, the fluridone concentration in the water is reduced through dilution due to water movement, uptake by plants, adsorption to the sediments, and break down from light and microbial action.

There are two major degradation products from fluridone: n-methyl formamide (NMF) and 3-trifluoromethyl benzoic acid. NMF has not been detected in studies of field conditions, including those at the maximum label rate.

Fluridone residues in sediments reach a maximum in one to four weeks after treatment and decline in four months to a year depending on environmental conditions. Fluridone adsorbs to clay and soils with high organic matter, especially in pellet form, and can reduce the concentration of fluridone in the water. Adsorption to the sediments is reversible; fluridone gradually dissipates back into the water where it is subject to chemical breakdown.

Impacts on Fish and Other Aquatic Organisms

Fluridone does not appear to have any apparent short-term or long-term effects on fish at application rates.

Fish exposed to water treated with fluridone absorb fluridone into their tissues. Residues of fluridone in fish decrease as the herbicide disappears from the water. The EPA has established a tolerance for fluridone residues in fish of 0.5 parts per million (ppm).

Studies on Fluridone's effects on aquatic invertebrates (i.e. midge and water flea) have shown increased mortality at label application rates.

Studies on birds indicate that fluridone would not pose an acute or chronic risk to birds. No studies have been conducted on amphibians or reptiles.

Human Health

The risk of acute exposure to fluridone would be primarily to chemical applicators. The acute toxicity risk from oral and inhalation routes is minimal. Concentrated fluridone may cause some eye or skin irritation. No personal protective equipment is required on the label to mix or apply fluridone.

Fluridone does not show evidence of causing birth defects, reproductive toxicity, or genetic mutations in mammals tested. It is not considered to be carcinogenic nor does it impair immune or endocrine function.

There is some evidence that the degradation product NMF causes birth defects. However, since NMF has only been detected in the lab and not following actual fluridone treatments, the manufacturer and EPA have indicated that fluridone use should not result in NMF concentrations that would adversely affect the health of water users. In the re-registration assessment that is currently underway for fluridone, the EPA has requested additional studies on both NMF and 3-trifluoromethyl benzoic acid.

For Additional Information

Environmental Protection Agency
Office of Pesticide Programs
www.epa.gov/pesticides

Wisconsin Department of Agriculture, Trade, and Consumer Protection
http://datcp.wi.gov/Plants/Pesticides/

Wisconsin Department of Natural Resources
608-266-2621
http://dnr.wi.gov/lakes/plants/

Wisconsin Department of Health Services
http://www.dhs.wisconsin.gov/

National Pesticide Information Center
1-800-858-7378
http://npic.orst.edu/

Hamelink, J.L., D.R. Buckler, F.L. Mayer, D.U. Palawski, and H.O. Sanders. 1986. Toxicity of Fluridone to Aquatic Invertebrates and Fish. Environmental Toxicology and Chemistry 5:87-94.

Fluridone ecological risk assessment by the Bureau of Land Management, Reno Nevada: http://www.blm.gov/pgdata/etc/medialib/blm/wo/Planning_and_Renewable_Resources/veis.Par.91082.File.tmp/Fluridone%20Ecological%20Risk%20Assessment.pdf

Project Title:

USE OF THE REGISTERED AQUATIC HERBICIDE FLURIDONE (SONAR”)
IN THE STATE OF NEW YORK

GENERIC ENVIRONMENTAL
IMPACT STATEMENT

Prepared for:

SePRO Corporation
11550 North Meridian Street
Carmel, Indiana 46032

Prepared by:

McLaren/Hart Environmental Engineering Corporation
25 Independence Boulevard
Warren, New Jersey 07059

April 19, 1995

Version 2.0

**** NOTE ****

This Generic Environmental Impact Statement is an extraction from the Final Generic Environmental Impact Statement, Use of the Registered Aquatic Herbicide Fluridone (Sonar®) and the Use of the Registered Aquatic Herbicide Glyphosate (Rodeo® and Accord®) in the State of New York, Version 5.0, dated January 10, 1995. The following document is intended for information purposes only, either in the State of New York, or elsewhere. It cannot be utilized in the development of a permit application for the use of Sonar® in the waters of the State of New York. Interested parties must reference the Final Generic Environmental Impact Statement, Use of the Registered Aquatic Herbicide Fluridone (Sonar®) an the Use of the Registered Aquatic Herbicide Glyphosate (Rodeo® and Accord®) in the State of New York, Version 5.0, dated January 10, 1995, as approved by the New York State Department of Environmental Conservation on January 25, 1995 for any permit applications for the use of Sonar® in waters of the State of New York.

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EXECUTIVE SUMMARY

This Generic Environmental Impact Statement was extracted from the Final Generic Environmental Impact Statement, Use of the Registered Aquatic Herbicide Fluridone (Sonar®) and the Use of the Registered Aquatic Herbicide Glyphosate (Rodeo® and Accord®) in the State of New York, Version 5.0, dated January 10, 1995. That Generic Environmental Impact Statement (GElS) was submitted to the New York State Department of Environmental Conservation (NYSDEC) on behalf of DowElanco and SePRO, in part, for the aquatic herbicide fluridone (Sonar®)[1]. It is the purpose of the GEIS to objectively evaluate the scientifically documented evidence regarding all aspects of the use of (Sonar®) for the control of nuisance aquatic weeds in waters of the State of New York. This document is intended to present a general description of the potential positive and negative impacts from the use of this product within waters of the State of New York.

The GEIS was prepared in accordance with 6 NYCRR Part 617, the New York State Environmental Quality Review Act (SEQR). The purpose of SEQR is to incorporate the consideration of environmental factors into the existing planning, review and decision-making processes of State, regional and local government agencies at the earliest possible time. An action is subject to review by NYSDEC under SEQR if any state or local agency has the authority to issue a permit or other type of approval over that action.

NYSDEC issued a Positive Declaration (as defined in § 617.10(b)) stating that any permits developed for the potential use of the Sonar® in the State of New York warrants a review under the SEQR process. As described in Section 1.2 of this GEIS, DowElanco and SePRO chose to prepared the Final Generic Environmental Impact Statement, Use of the Registered Aquatic Herbicide Fluridone (Sonar®) and the Use of the Registered Aquatic Herbicide Glyphosate (Rodeo® and Accord®) in the State of New York, Version 5.0, dated January 10, 1995, as described in § 617.15 to facilitate the development of individual permits for potential users of those products. Section 617. 15 (a)(4) allows for the development of a GEIS to assess the potential environmental effects of an entire program or plan having wide application.

The preparation of this GEIS is intended to provided potential users and interested parties with information specific for Sonar® and its positive and negative impacts on surface water resources of New York State.

The U.S. Environmental Protection Agency (USEPA) approved the label for Sonar® on March 31, 1986. The USEPA registration number for Sonar® A.S. is 62719-124. The USEPA registration number for Sonar® SRP is 62719-123. DowElanco received New York State registration approval for Sonar® SRP on February 9, 1993. DowElanco applied for, and was

[1]The rights of the trademarked product Sonar® were purchased by the SePRO Corporation of Carmel, Indiana from DowElanco of Indianapolis, Indiana. The Department of Environmental Conservation has approved the application to change just the name on the labels of Sonar® A.S. and Sonar® SRP. The revised labels are identical with DowElanco's name replaced by SePRO.

granted, a Special Local Needs (SLN) registration for Sonar A.S.® for the control of Eurasian watermilfoil (Myriophyllum spicatum L.), at application rates of *50* ppb or less in freshwater ponds, lakes, and reservoirs The SLN registration was received by DowElanco on February 9, 1993. The SLN registration number is SLN NY-930001.

The proposed action is the use of the aquatic herbicide Sonar® for the control of nuisance aquatic vegetation in waterbodies located in the State of New York. The use of the products can be an important component of a comprehensive management approach to limiting the production and spread of certain aquatic macrophytes. These macrophytes are often undesirable, opportunistic introduced species. These species can become a nuisance as a result of the production of excessive biomass or because of the growth habits or physical architecture of the plant. The production of these plant species can reduce the recreational use of a waterbody by interfering with swimming, boating, or fishing, They may also clog intake screens and turbines, impart an unpleasant taste to the water, reduce the presence of native aquatic species, and modify the aquatic habitat for indigenous organisms.

Because of its mat forming characteristics, excessive growth of Eurasian watermilfoil (a primary target species for Sonar®) may also present a safety hazard to the recreational use of a waterbody. The mats may conceal rocks, logs and other obstructions that could damage moving boats or injure skiers. Additionally, the mats may entangle swimmers, potentially resulting in drowning. Drowning as a result of entanglement in Eurasian watermilfoil mats have been documented in New York and Michigan.

Sonar is a systemic aquatic herbicide produced by SePRO. Sonar® works by interrupting the photosynthetic abilities of the target plants. Specifically, Sonar® inhibits the formation of the accessory pigment carotene within the target plants. In the absence of carotene, chlorophyll is rapidly degraded by sunlight, thereby preventing the formation of carbohydrates necessary to sustain the plant.

The active ingredient in Sonar® is fluridone (1-methyl-3-phenyl-5-[3-(trifluoromethyl)phenyl]-4[1H]-pyridinone). The U.S. Environmental Protection Agency (USEPA) Shaughuessy code for fluridone is 112900-6. Sonar® is packaged in two formulations: Sonar® SRP and Sonar® A.S. Sonar® SRP is a pelleted formulation containing 5 % fluridone. Sonar® SRP is generally applied via broadcast spreading. Sonar® A.S. is a liquid formulation that is mixed with water prior to application. Sonar® A.S. is generally applied via broadcast surface spraying or through the use of underwater hoses.

For both Sonar® formulations, the critical feature with regard to aquatic macrophyte control is obtaining an adequate concentration of the product in the treated area for a sufficient time period to produce the effect. Under optimum conditions the desired level of target aquatic macrophyte control is achieved 30-90 days after the use of Sonar®. Sonar® is absorbed from water by plant shoots and from the hydrosoil by the roots of aquatic vascular plants

The Milfoil Study Committee of the Vermont Department of Environmental Conservation (VDEC) reported that the VDEC has been attempting to control the spread of Eurasian watermilfoil through non-chemical means since 1978. The primary means have been mechanical harvesters and bottom barriers. Despite the attempts, the Committee has noted that Eurasian watermilfoil has continued to spread within infected lakes and to uninfested lakes. The Study Committee recommended to the VDEC in 1993 that they use aquatic herbicides on a site-specific basis for the control of introduced, exotic vascular aquatic plant species in Vermont. The Committee does not recommend the use of Diquat or Endothall because their use would not meet the statutory requirement of pesticide minimization in a long-range management plan and they do not recommend the use of 2,4-D because of the uncertainty about potential human health effects.

It is the aim of this document to evaluate the role of Sonar® in the management of aquatic nuisance vegetation and the potential for impacts from that use. This Generic Environmental Impact Statement evaluates Sonar® with respect to the following issues:

- Environmental Setting
- General Description of Sonar® and Its Active Ingredients
- Significant Environmental Impacts
- Potential Public Health Impacts
- Mitigation Measures
- Unavoidable Environmental Impacts
- Alternatives

This Final Generic Environmental Impact Statement (GEIS) consists of:

- The text of the FGEIS as amended from the Draft Generic Environmental Impact Statement (DGEIS), based on comments received by the Department;
- The Written Comments received on or before the close of the public comment period on June 6, 1994 and the responses to those comments are contained in Appendix G to this document;
- The Hearing Comments as received at the May 4, 1994 Hearing in Lake George; the May 5, 1994 Hearing in Poughkeepsie; and the May 11, 1994 Hearing in Rochester and the responses to those comments are contained in Appendix H to this document.

The DGEIS was accepted as complete on April 6, 1994 and available for public comment for 60 days until June 6, 1994. There were 3 public hearings held as fo1lows

- May 4, 1994 at 7:00 pm in the Lake George Town Center on Old Post Road in the Village of Lake George;

- May 5, 1994 at 7:00 pm in the Best Western Inn and Conference Center at 679 South Road (Route 9) in the City of Poughkeepsie; and

- May 11, 1994 at 7:00 pm in the Marriot Thruway at 5257 West Henrietta Road in the City of Rochester.

1.0 INTRODUCTION

1.1 PURPOSE OF THE GENERIC ENVIRONMENTAL IMPACT STATEMENT (GEIS)

This Generic Environmental Impact Statement was extracted from the Final Generic Environmental Impact Statement Use of the Registered Aquatic Herbicide Fluridone (Sonar®) and the Use of the Registered Aquatic Herbicide Glyphosate (Rodeo® and Accord®) in the State of New York Version 5.0, dated January 10, 1995. That Generic Environmental Impact Statement (GEIS) was submitted to the New York State Department of Environmental Conservation (NYSDEC) on behalf of DowElanco and SePRO, in part, for the aquatic herbicide fluridone (Sonar®)[1]. It is the purpose of the GEIS to objectively evaluate the scientifically documented evidence regarding all aspects of the use of (Sonar®) for the control of nuisance aquatic weeds in waters of the State of New York. This document is intended to present a general description of the potential positive and negative impacts from the use of this product within waters of the State of New York.

1.2 OBJECTIVE OF THE GEIS

The development of the Final Generic Environmental Impact Statement, Use of the Registered Aquatic Herbicide Fluridone (Sonar®) and the Use of the Registered Aquatic Herbicide Glyphosate (Rodeo® and Accord®) in the State of New York Version 5.0, dated January 10**,** 1995 provided potential users of those products with a general understanding of the various results that might be associated with the use of Sonar® in the waters of the State of New York. By developing the GEIS, SePRO has provided the information necessary for individual potential applicators to easily develop the necessary permit applications. However, the approach taken through the development of this GEIS is not intended to prevent any applicant from preparing a site specific supplement to the Final Programmatic Environmental. Impact Statement on Aquatic Vegetation Control (NYSDEC, l981a) in the development of a permit for the use of fluridone (Sonar®) in surface waters of New York State.

The preparation of this GEIS is intended to provided potential users and interested parties with information specific for Sonar® and its positive and negative impacts on surface water resources of New York State.

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[1]The rights of the trademarked product Sonar® were purchased by the SePRO Corporation of Carmel, Indiana from DowElanco of Indianapolis, Indiana. The Department of Environmental Conservation has approved the application to change just the name on the labels of Sonar® A.S. and Sonar® SRP. The revised labels are identical with DowElanco's name replaced by SePRO.

1.3 REGULATORY FRAMEWORK

The GEIS was prepared in accordance with 6 NYCRR Part 617, the New York State Environmental Quality Review Act (SEQR). The purpose of SEQR is to incorporate the consideration of environmental factors into the existing planning, review and decision-making processes of State, regional and local government agencies at the earliest possible time. An action is subject to review by the NYSDEC under SEQR if any state or local agency has the authority to issue a permit or other type of approval over that action.

NYSDEC issued a Positive Declaration (as defined in § 617.10(b)) stating that any permits developed for the potential use of the Sonar® in the State of New York warrant a review under the SEQR process. As described in Section 1.2 of this GEIS, DowElanco and SePRO chose to prepare the Final Generic Environmental Impact Statement, Use of the Registered Aquatic Herbicide Fluridone (Sonar®) and the Use of the Registered Aquatic Herbicide Glyphosate (Rodeo® and Accord®) in the State of New York, Version 5.0, dated January 10, 1995, as described in § 617.15 to facilitate the development of individual permits for potential users of the products. Section 617.15 (a)(4) allows for the development of a GEIS to assess the potential environmental effects of an entire program or plan having wide application.

The regulations concerning the use of pesticides in NYS are defined in 6 NYCRR Part 325 through 327. The regulations addressing the use of pesticides in wetlands are defined in 6 NYCRR Part 663 and within the Adirondack Park, 9 NYCRR Part 578.

The USEPA issued an Experimental Use Permit (No. 1471-EUP-67) for Sonar® in 1981. The USEPA approved the label for Sonar® on March 31, 1986. The USEPA registration number for Sonar® A.S. is 62719-124. The USEPA registration number for Sonar® SRP is 62719-123. DowElanco received New York State registration approval for Sonar® SRP on February 9, 1993. DowElanco applied for, and was granted, a Special Local Needs (SLN) registration for Sonar® A.S. for the control of Eurasian watermilfoil (Myriophyllum spicatum L.), at application rates of 50 ppb or less in freshwater ponds, lakes, and reservoirs. The SLN registration was received by DowElanco on February 9, 1993. The SLN registration number is SLN NY-930001. Pursuant to the registration conditions described in 6 NYCRR Part 326, fluridone may only be used as follows:

1. Application of aqueous suspension formulations are permitted in water of the State at application rates not to exceed 50 ppb of the active ingredient fluridone.

2. Application of pellet formulations are not permitted in waters less than two feet deep.

3. Swimming is not allowed in treated areas for a period of 24 hours following the application.

Sonar® is registered for use without restrictions in all states bordering New York State. Sonar® is not registered in Canada. The approved labels and Material Safety Data Sheets (MSDS) for Sonar® SRP and Sonar® A.S. are presented in Appendix A.

1.4 IDENTIFICATION AND JURISDICTION OF THE INVOLVED AGENCIES

The following agencies were identified as involved agencies for purposes of the development of this GEIS:

a. New York State Department of Environmental Conservation (NYSDEC) - Responsible for implementation of the laws and regulations pertaining to the management of environmental resources for the State of New York
b. New York State Department of Health (DOH) - Responsible for potential public health issues associated with the use of

the Products.
c. New York State Office of General Services (OGS) - Responsible for the management of property owned by the State of New York. As pertaining to this project, they are responsible for the management of the lakes and/or lake bottoms owned by the State of New York.
d. Adirondack Park Agency (APA) - responsible for implementation of the Adirondack park Land Use and Development Plan (as described by the Adirondack Park Agency Act).
e. New York State Department of State (DOS) - Responsible for the administration of the Coastal Zone Program.

By agreement of the involved agencies, NYSDEC designated as the lead agency for the GEIS.

2.0 DESCRIPTION OF THE PROPOSED ACTION - USE OF SONAR®

The proposed action is the use of the aquatic herbicide Sonar® for the control of nuisance aquatic vegetation in waterbodies located in the State of New York. The use of Sonar® can be an important component of a comprehensive management approach to limit the spread of certain aquatic macrophytes. These macrophytes can be undesirable in certain circumstances. They may be introduced species, which because of the lack of controlling ecological factors, reach a nuisance stage in terms of extreme numbers or biomass. Such exponential growth can reduce the recreational use of a waterbody by interfering with swimming, boating, or fishing. They may also clog intake screens and turbines, impart an unpleasant taste to the water, and reduce the presence of native aquatic species (Madsen et al., 1991a). Vermont Department of Environmental Conservation notes that nuisance vegetation may modify the aquatic habitat for indigenous organisms (VDEC, 1993).

Because of its mat forming characteristics, excessive growth of Eurasian watermilfoil (a primary target species for Sonar®) may also present a safety hazard to the recreational use of a waterbody. The mats may cover rocks, logs, and other obstructions that could damage moving boats or injure water skiers. Additionally, the mats may entangle swimmers, potentially resulting in drowning. Drowning as a result of entanglement in Eurasian watermilfoil mats have been documented in New York (Long et al., 1987) and Michigan (COLAM, 1992).

2.1 GENERAL DESCRIPTION OF THE AQUATIC HERBICIDE FLURIDONE (SONAR®)

Sonar® is a systemic aquatic herbicide produced by SePRO. Sonar® works by interrupting the photosynthetic abilities of the target plants. Specifically, Sonar® inhibits the formation of the accessory pigment carotene within the target plants. In the absence of carotene, chlorophyll is rapidly degraded by sunlight, thereby preventing the formation of carbohydrates necessary to sustain the plant.

The active ingredient in Sonar® is fluridone (1 -methyl-3-phenyl-5-[3-(trifluoromethyl)phenyl]- 4[1H]-pyridinone). The U.S. Environmental Protection Agency (USEPA) Shaughnessy code for fluridone is 112900-6. Fluridone was registered with the USEPA in 1986 and with the NYSDEC in 1993. Sonar® is packaged in two formulations: Sonar® SRP (Slow Release Pellets) and Sonar® A.S. (Aqueous Suspension). Sonar® SRP is a pelleted formulation containing 5% fluridone. Sonar® SRP is generally applied via broadcast spreading. Sonar® A.S. is a liquid formulation that is mixed with water prior to application. Sonar® A.S. is generally applied via broadcast surface spraying or through the use of underwater application equipment.

For both formulations, the critical feature with regards to aquatic macrophyte control is obtaining an adequate concentration of the product in the treated area for a sufficient time period to produce the effect. Under optimum conditions, as noted on the approved label, the desired level of target aquatic macrophyte control is achieved 30 - 90 days after the application of

Sonar®. Sonar® is absorbed from water by plant shoots and from the hydrosoil by the roots of aquatic vascular plants.

2.1.1 Purpose of the Product

As a systemic aquatic herbicide, Sonar® is designed to control broad-leaved submerged aquatic macrophyte species. This includes nuisance species such as Eurasian watermilfoil and curlyleaf pondweed, as well as native pondweeds. As opposed to a non-selective contact herbicide which will kill any plant material that it comes in contact with. Sonar® is intended for a select group of target species, which are listed on the registered labels. Several plant species that perform valuable functions in the aquatic environment, mainly floating and emergent species such as algae, bulrush, pickerel weed, cattails and waterlilies, are either not impacted, marginally impacted, or are impacted in a positive manner by the use of Sonar® at the labeled application rates. It is noted that the target species for Sonar® also perform valued functions, though the level of function can be dependent on the density of the target species.

The product's manufacturer, supported by various researchers, believes that the selectivity of Sonar® to the intended group of broad-leaved submergent aquatic macrophytes can be focused based on the application rates. Those species such as Eurasian watermilfoil and curlyleaf pondweed which are highly sensitive to fluridone, can be treated at sufficiently low rates that those species which are not quite as sensitive will only be moderately impacted. However, it is understood that the higher the application rate, the broader the impacts become within that category of macrophytes which are considered as potential targets or are sensitive to fluridone.

It is for these reasons, that the authors believe that the use of the term "selective" is appropriate for a discussion of Sonar®. The authors have attempted to objectively present all available information with regard to questions of selectivity and varying responses based on observed application rates, which again, is the purpose of this document. This includes information on the observed rapid reestablishment of native plant communities within a growing season of Sonar® application. Kenaga (1993) states in his document that often there are other factors related to impacts to native aquatic plant communities which are not associated with the use of Sonar®. Of particular note is that the intended opportunistic target species may so dominate the plant community that the remaining non-target community is reduced and very poor. This is where the rapid reestablishment of the non-target community that is documented in other studies (and is discussed in the GEIS) would be of importance.

2.1-2 Need for the Product

Sonar® is all aquatic herbicide which is intended for the selective control of nuisance aquatic macrophytes. Sonar® is especially effective in controlling or removing Eurasian watermilfoil and curlyleaf pondweed. Eurasian watermilfoil is an exotic, invasive aquatic macrophyte that can significantly affect the littoral characteristics of a freshwater pond or lake (Pullman, 1993 and VDEC, 1993). VDEC (1993) reports that in Vermont the number of confirmed lake infestations by Eurasian watermilfoil has grown exponentially, from fewer than 5 in 1962, to more than 35 lakes in 1992. Eighty-Five percent of that growth has occurred since 1982 and has occurred despite the efforts of non-chemical control methods. Coalition of Lakes Against Milfoil (COLAM, 1992) notes that 10 counties in the State of New York had reported occurrences of Eurasian watermilfoil in 1980. They report that by 1992, that number had grown to 35 counties. In its 1993 Annual Report on the Aquatic Plant Identification Program, the Rensselaer Fresh Water Institute notes that 38 counties had documented populations of Eurasian watermilfoil in 1993 (Eichler and Bombard, 1994). As a result of the documented expansion in the occurrences of Eurasian watermilfoil throughout the State of New York, the need for environmentally sound, effective methods for control of this nuisance species is evident Westerdahl and Hall (1987) note that Eurasian watermilfoil is extremely susceptible to fluridone, (the active ingredient in Sonar®).

Curlyleaf pondweed is also an exotic species that has spread throughout the United States (NYSDEC, 1990), Pullman (1992) notes that the curlyleaf pondweed will thrive in most plant productive lakes and that it can be a severe nuisance during the early part of the peak recreational use period in lakes in the northern United States. Pullman (1992) reports that Sonar® was used selectively for the control of curlyleaf pondweed in lakes in Michigan.

2.1.3 Benefits of the Product

The use of Sonar® will allow for a comprehensive approach to the control and management of target aquatic macrophyte species. It allows for the selective control of target macrophyte species and for the restoration of native plant communities. Through the use of Eurasian watermilfoil management techniques, which include an aquatic herbicide such as Sonar®, the negative attributes of the growth of this nuisance weed can be reversed. Pullman (1993) reports that the use of Sonar® in lakes in Michigan has resulted in the removal of Eurasian watermilfoil and allowed for the restoration of the native plant community. At concentrations above 8 ppb, Sonar® has never failed to control the growth of both Eurasian watermilfoil and curlyleaf pondweed.

Based on an economic study conducted in the Okanagan Valley region of British Columbia, the British Columbia Ministry of Environment, Lands and Parks (BCMELP, 1991) noted that the failure to control Eurasian watermilfoil results in economic impacts to the area surrounding the affected waterbody. Their study was conducted-in an area containing seven mainstream lakes and one upper elevation lake, of which 1000 hectares of shoreline were reported to be infested with Eurasian watermilfoil. They estimated losses in several economic areas; including transportation, the restaurant industry, the accommodation sector, and the shopping sector. BCMELP (1991) projected that a no-action alternative to managing for Eurasian watermilfoil would result in a revenue loss of $85 million dollars in 1990 to the region (or 26.5% of 1989 revenues). BCMELP (1991) also predicted a loss of 1700 employment positions in the tourist industry and a loss in real estate values of $360 million in the region. However, the British Columbia Ministry of Environment, Lands and Parks has not verified these projected economic losses. The use of Sonar® as a management approach would help alleviate those concerns.

The use of Sonar®, as per the NYS registered labels, would allow for the alleviation of safety concerns brought about by the infestation of a lake by Eurasian watermilfoil. Eurasian watermilfoil can reach a stage where thick mats will form at the waters surface. Under these conditions, rocks, logs, and other obstructions will be concealed. These objects would damage moving boats or injure skiers attempting to pass through the matted areas. Sonar® could be used to remove the Eurasian watermilfoil, and allow for the safe recreational use of the lake.

Sonar® can be a selective means of managing nuisance aquatic vegetation. The benefit of its use is the selective removal, of those exotic aquatic macrophytes considered to be a nuisance to the use, function and value of a lake, while allowing for the reestablishment of more valuable native plant species.

2.1.4 History of Product Use

The USEPA issued an Experimental Use Permit (No. 147l-EUP-67) for Sonar® in 1981. The USEPA approved the label for Sonar® on March 31, 1986. There were no use restrictions included for treated ponds (waterbodies 10 acres or less in size). For treated lakes and reservoirs, the only restriction was the prohibition on the use of Sonar® within 1/4 mile (1320 feet) of any potable water intake. There were no restrictions on uses of treated water, Sonar® and its active ingredient, fluridone, are registered only for aquatic uses. Specifically, it is registered for the management of aquatic vegetation in freshwater ponds, lakes, reservoirs, drainage canals and irrigation canals. The Sonar® SRP formulation is also registered for application to rivers. The USEPA registration number for Sonar® A.S. is 62719-124. The USEPA registration number for Sonar® SRP is 62719-123. DowElanco received New York State registration approval for Sonar® SRP on February 9, 1993. DowElanco applied for, and was granted, a Special Local Needs (SLN) registration for Sonar® A.S. for the control of Eurasian watermilfoil (Myriophyllum spicatum L.), at application rates of 50 ppb or less in freshwater ponds, lakes, and reservoirs. The SLN registration was received by DowElanco on February 9, 1993. The SLN registration number is SLN NY-930001.

Pursuant to the registration conditions described in 6 NYCRR Part 326, fluridone may only be used as follows:

1. Application of aqueous suspension formulations are permitted in water of the State at application rates not to exceed 50 ppb of the active ingredient fluridone.

2. Application of pellet formulations are not permitted in waters less than two feet deep.

3. Swimming is not allowed in treated areas for a period of 24 hours fol1owing the application.

Sonar® herbicides have been used primarily for the control of submersed nuisance aquatic plants, primarily hydrilla (Hydrilla verticillata) in the southern states, and Eurasian watermilfoil in the northern United States (U.S.). Curlyleaf pondweed (Potmogeton crispus) is also frequently a target species of aquatic plant management programs. Applications have provided successful management of target species, with control lasting from one to several seasons after treatment.

Lack of satisfactory control within treated areas is generally evident only where moderate to rapid rates of water exchange cause rapid dilution of fluridone treated water, resulting in too little contact time with target plants for adequate herbicide uptake. This situation may occur at water inlets into otherwise quiescent waterbodies.

Experience during the years since registration has shown that the use of Sonar® A.S. in treating water at concentrations that are lower than those listed on the Federal label can provide excellent control of Eurasian watermilfoil (Pullman, 1992). This is especially true in situations where treatments can be applied to whole water bodies and there is limited opportunity for dilution with untreated water. The low use rate experience made possible a 24(c) Special Local Need registration of Sonar® A. S. in NYS for control of Eurasian watermilfoil using reduced treatment rates.

Sonar® applications for control of Eurasian watermilfoil in northern, states have been made most frequently in Michigan. Applications have also been made in Indiana, Illinois, Minnesota, New Jersey and Washington. As indicated, these treatments have provided excellent control of target plants. Reduced Sonar rates, early season treatments, and uniform product applications over the area to be treated have removed nuisance growths of Eurasian watermilfoil, while minimizing the herbicide impacts on non-target species, including other aquatic plants listed on the Sonar® labels as species controlled.

2.1 4.1 Registration Status in States and Canadian Provinces That Are Neighboring New York State

Sonar® is registered, without any use restrictions, in Pennsylvania, New Jersey, Connecticut, Massachusetts, and Vermont. Furthermore, there are no restrictions on the use of Sonar® herbicides in any other state in which it is registered. Sonar® herbicides are not registered for use in Canada. No registration actions have been submitted to Canada.

2.2 GENERAL LOCATION OF THE PROPOSED ACTION

For the purposes of this portion of the GEIS, the general location for the proposed action is in the surface waters of the State of New York. The proposed action is the use of the aquatic herbicide Sonar® for the control of certain nuisance aquatic macrophytes. A specific description of the actual body of water in which Sonar® is intended for use would be included in the individual permit applications. Sonar® A.S. is registered in New York for use in freshwater lakes, ponds, and reservoirs. Sonar® SRP is registered for use in freshwater lakes, ponds, reservoirs, drainage canals, irrigation canals, and rivers. Under Article 24 of the Environmental Conservation Law, some ponded water may be described as wetlands.

NYSDEC (1987) reports that over 7500 lakes, ponds, and reservoirs can be found in the State of New York. While NYSDEC (1990) states that there are no scientific terms for the three types of waterbodies, it notes that ponds are generally small, shallow waterbodies with little or no wave action, that usually exhibit uniform temperature distributions. Lakes are generally large and deep water bodies that exhibit periodic thermal stratification and may have rocky, wave-impacted shorelines due to exposure to prevailing winds. Water in the lake is contributed from the surrounding land which is termed the water basin. Water can be contributed to the lake through streams, rivers, groundwater, or general surface runoff. Reservoirs are man-made lakes. For purposes of label interpretation, Sonar® labels define a pond as a body of water 10 acres or less in size. A lake or reservoir is defined as greater than 10 acres in size.

2.3 POTENTIAL AQUATIC MACROPHYTE TARGET SPECIES

This GEIS is a supplement to the NYS Environmental Impact Statement, dated May 1, 1981 (NYSDEC, l981a). Based on the registered label for Sonar® SRP, the aquatic macrophyte species listed in this section are considered to be potential target species for this product. However, not all of the aquatic macrophyte species described on the product labels are found in the State of New York. The detailed discussions of the target species are limited to those species indigenous to New York State. With respect to Sonar® A.S., it should be noted that this product is registered in NYS only for the treatment of Eurasian watermilfoil. However, at the registered application rate for Sonar® A.S., the plants in the following sections would be expected to be either affected, or not affected, depending on the species sensitivity to fluridone.

2.3.1 Eurasian Watermilfoil (Myriophyllum spicatum L.)

A primary target species for Sonar® in New York State is Eurasian watermilfoil (Myriophyllum spicatum L.) Eurasian watermilfoil is an aquatic plant found in the taxonomic family Haloragaceae. It is a rooted, vascular submergent macrophyte with long stems and feathery perennial leaves. Plants form no specialized overwintering vegetative structures such as turions. The leaves are generally attached along the entire stem in whorls of four and can be in excess of 35 mm in length. Each leaf is composed of 7 to18 pairs of leaflets (Pullman, 1993). The leaflets are mostly straight and of equal length. The inflorescence is terminal and extends above the water surface. Upper flowers are generally staminate. Lower flowers are generally pistillate (Britton and Brown, 1970b). Eurasian watermilfoil is an invasive, opportunistic exotic plant that is native to Europe, Asia, and North Africa (Pullman, 1993 and Long et al., 1987). Hotchkiss (1972) reports that Eurasian watermilfoil is distributed across the northern tier of the United States, from California to Vermont.

2.3.2 Other Potential Aquatic Macrophyte Target Species

The following species are listed on the federally registered labels for Sonar® A.S. and Sonar® SRP as potential species targeted for control. These species are consistent with those species listed on the New York registered label for Sonar® SRP. Sonar® A.S. is only registered in the State of New York for the management of Eurasian watermilfoil. The selection of A.S. versus SRP is further addressed in Section 4.2 and 4.5. Only those potential target species actually occurring in New York State are discussed in this section. Species listed in Table 2-1 are found on the New York registered Sonar® SRP label, but do not occur in New York State.

TABLE 2-1

AQUATIC MACROPHYTES LISTED ON THE REGISTERED LABELS FOR SONAR® BUT NOT FOUND IN THE STATE OF NEW YORK

Alligatorweed (Alternanthera philoxeroides)
Giant Cutgrass (Zizaniopsis miliacea)
Hydrilla (Hydrilla vertilillata)

The following potential target species are noted as being either controlled or partially controlled, consistent with the Sonar® SRP label. The controlled notation indicates that the plant species would be removed from the treatment area by the use of fluridone at the application rate labeled for Sonar® SRP in NYS. The partially controlled notation indicates that at the 50 ppb maximum application rate for Sonar® A.S. and at the maximum label application rate for Sonar® SRP in NYS, some herbicidal effects or growth suppression would be observed on the plant. The level of herbicidal effects, however, would not be such that the species would be removed from the waterbody and a claim for commercial control of the macrophyte could be maintained, Plant distributions in this section are based on Hotchkiss (1972), Mitchell (1986), Magee (1981), Tiner (1987) and ACOE (1977).

Submerged, Floating-leaved, and Floating Plants:
American Lotus (Nelumbo lutea) [Partially controlled]

The American lotus or yellow lotus is found in the taxonomic family Nymphaeaceae. This plant is listed as a rare native plant species in NYS. The lotus is characterized by grayish-green leaves which are as much as 2 feet across and float or stand above the water.

Bladderwort (Ultricularia spp.) [Controlled]

Bladderworts are found in the taxonomic family Lentibulariaceae. Magee (1981) reports that bladderworts are generally found in ponds, shallow lakes and sluggish streams, up to 1.2 meters in depth. Bladderworts are long, slender, free-floating plants with finely forked leaves, bearing small air bladders in the forks of the divisions. When treated at low Sonar® rates for control of Eurasian watermilfoil, bladderwort species will increase in area covered after the treatment (Pullman, 1993).

Common Coontail (Ceratophyllum demersum) [Controlled]

Coontail, or hornwort, is found in the taxonomic family Ceratophyllaceae. NYSDEC (1990) reports that coontail is a free-floating perennial which lacks roots. The stems are generally slender, and hollow and can grow up to 50 cm in length. Leaves are submersed and whorled in groups of 5 to 12 and are abundantly located near the stem tip. The primary method for coontail reproduction is through fragmentation. When treated at low Sonar® rates used for watermilfoil, coontail displays temporary herbicidal symptoms (Pullman, 1993).

Common Elodea (Elodea canadensis) [Controlled]

Common elodea, or ditch-moss, is found in the taxonomic family Hydrocharitaceae. NYSDEC (1990) notes that common eloclea s a submersed perennial, with thin, branched green stems. It often forms large masses near the bottom. Leaves are arranged in whorls of three or are opposite. Leaves are generally 10 to 13 mm long and 3 to 5 mm wide. Elodea is considered to be an aquatic nuisance species (Nichols and Shaw, 1986). Elodea grows on a wide variety of sediments, though it grows best on fine sediments where organic matter ranges from 10% to 25%. Elodea overwinters as an entire plant under the ice and grows quickly in the spring from the dormant stem apices. As with Eurasian watermilfoil, elodea spreads primarily through the disposal of stem fragments. Elodea is considered to be an important substrate for invertebrates. It is not considered to be important for invertebrates as a food source or as a place to lay eggs. Elodea has been noted to inhibit the growth of phytoplankton in a waterbody (Nichols arid Shaw, 1986).

Egeria, Brazilian Elodea (Egeria densa) Controlled

Egeria is found in the taxonomic family Hydrocharitaceae, This plant is an exotic species that is listed in NYS as a rare, escaped plant species,

Fanwort (Cabomba caroliniana) Controlled

Fanwort is an exotic introduced species introduced to NYS. It is found in the taxonomic family Nymphaeaceae. It is a submersed, floating perennial herb that is often rooted. The stems are slender and the leaves are opposite and whorled. Flowers appear above the upper leaves and are usually white or pink,

Naiad (Najas spp.) Controlled

Plants in this family, Najadaceae, are distributed from Newfoundland and Quebec to Minnesota, and south to Florida, They are generally found in shallow, quiet waters of ponds, lakes, pools, and sluggish streams. Magee (1981) notes that these plants arc slender, with many-branched stems up to 1 meter long. The leaves are opposite, slender and thread-like. Flowers are small and inconspicuous. Naiad (Najas quadalupensis var. olivacea) and holly-leaved naiad or maxine naiad (Najas marina), are listed as rare native plants in NYS.

Parrotfeather (Myriophyllum brasiliense) Partially Controlled

Hotchkis (1972) notes that parrotfeather is a common aquarium plant that is originally from South America. Parrotfeather is found in inland freshwater marshes and ponds.

Pondweed (Potamogeton spp.) Controlled

The pondweed family, Potamogetonaceae, is distributed from Newfoundland and Quebec to southern Alaska, south from Florida to Ca1ifonia. Pondweeds are generally found in still waters of ponds, lakes to moderately moving streams and rivers. Magee (1981) reports that pondwceds have slender, flexible, underwater stems bearing variable leaves in two vertical rows and opposite, elliptic floating leaves. Flowers are borne on spikes above the water surface. One species of pondweed (Ogdens's pondweed, Potamogeton ogdenii) is listed as an endangered native species in NYS. Hill's pondweed (P. hillii) is listed as a threatened native plant species in NYS. Pondweed (P. confervoides), northern pondweed (P. alpinus) and sheathed pondweed (P. filiformis var. occidentalis) are listed as rare native plant species in NYS.

Curlyleaf pondweed (Potamogeton crispus) is an exotic species that is considered to be a nuisance aquatic weed. Nichols and Shaw (1986) note that curlyleaf pondweed is native to Eurasia. It overwinters under the ice and its primary mode of spread is through the dispersal of dormant apices or turions. It prefers a water depth of one to three meters and a fine sediment texture with 10% to 25% organic content. It will survive in highly eutrophic conditions. Curlyleaf pondweed will form dense surface mats, which disrupt native plant communities, Spatterdock (Nuphar sp.) Partially Controlled

Spatterdock (Family Nymphaeaceae) is found in inland and coastal, fresh water marshes, ponds, lakes, pools, and the borders of slowly moving streams. Leaves vary greatly in size, but are generally large and lance-like in shape. In the form of the species indigenous to the northeastern United States, the leaves generally float on the surface of the water (Hotchkiss, 1972). Low Sonar® application rates used for treatment of Eurasian watermilfoil do not control spatterdock, but may produce temporary herbicidal effects.

Waterlily (Nymphaea spp.) Partially Controlled

Waterlilies (Family Nymphaeaceae) are aquatic herbs with thick cylindric, horizontal rootstocks. The leaves are generally large and cordate. Flowers are showy (Britton and Brown, 1970b). Waterlillies are found in slow, standing water in ponds, lakes or slowly moving treams. The three species of waterlily commonly found in New York State include Nymphaea odorata, N. tuberosa, and N. alba. Low Sonar® application rates used for treatment of Eurasian watermilfoil do not control waterlily, but may produce temporary herbicidal effects (Pullman, 1993).

Watermilfoil (Myriophyllum spp.) Controlled

Native species of Myriophyllum (Family Haloragaceae) are submersed, stout-stemmed perennials (Fairbrothers and Moul, 1965). There are generally 5 to 13 pairs of leaflets per leaf with each leaf approximately 4 cm long. Flowers are small and inconspicuous and occur in the axils of the upper leaves. Watermilfoil is found in ponds, lakes, sluggish streams, and shorelines. Eurasian watermilfoil (Myriophyllum spicatum L.) is considered to be an exotic nuisance weed (Nichols and Shaw, 1986).

Watermilfoil (Myriophyllum alterniflorum) is listed as a rate native plant species in NYS.

Waterprimrose (Ludwigia spp.) Partially Controlled

Waterprimroses are found in the evening-primrose family (Onagraceae). Plants in the genus Ludwigia are perennial or annual herbs, with alternate, usually entire leaves. They are generally found in freshwater marshes (Britton and Brown, 1970b). Ludwigia (Lugwigia sphaerocarpa) is listed as a rare plant species in NYS. Low Sonar® application rates used for treatment of Eurasian watermilfoil will only produce temporary herbicidal effects in waterprimrose

Waterpurslane (Lugwigia palustris) Partially Controlled

Waterpurslane, or false loosestrife (Family Onagraceae), is found along streams or springy areas. It can be found partly or wholly submerged in shallow water or sprawling over mud (Magee, 1981). It is a plant with a prostrate stem, with rooting occurring at the lower and middle nodes. Waterpurslane will often form mats. The leaves of the species are opposite and entire. The flowers of the species are small and found in the leaf axils. Low Sonar® application rates used for the treatment of Eurasian watermilfoil will only produce temporary herbicidal effects in waterpurslane.

Watershield (Brasenia schreberi) Partially Controlled

Watershield (Family Cabombaceae) is found in ponds, lakes, pools, and margins of slowly moving streams. It is found in water up to 1.2 meters in depth. The plant has floating leaves and flowers attached to flexible underwater petioles which are connected to thick rhizomes embedded horizontally in the mud. The leaves are large; growing up to 25 cm. The flowers are pinkish, with dark red centers. Low application rates used for the treatment of Eurasian watermilfoil will only produce temporary herbicidal effects in watershield.

Emergent and Marginal Plants:

Reed Canarygrass (Phalaris arundinaceae)
Reed canarygrass (Family Poaceae) is a grass that grows up to 2 meters in height. It is primarily found in marshes, wet meadows, and in ditches (Magee, 1981). Reed canarygrass normally grows in dense colonies. The leaf blades are long (up to 3.6 meters) and flowers are borne in a narrow, dense panicle. Reed carnarygrass is not controlled by Sonar at low Eurasian watermilfoil treatment rates.

Smartweed, Pennsylvania (Polygonum pensylvanicum)
The forms of species within this genus (Family Polygonaceae) are highly variable. Leaves are generally lance-like. The flowers are rose-pink or white. Pennsylvania smartweed is found in damp soil, roadsides, or fields (Peterson and McKenney, 1968).

Smartweed is not controlled by Sonar® at the low concentrations used to treat Eurasian watermilfoil.

Smartweed, swamp (Polygonum coccineum)

The forms of species within this genus are highly variable. This species has an erect form in the terrestrial environment and an aquatic form with floating leaves. Leaves are lance-like. Flowers are showy and pink.

Swamp smartweed is found in swamps and in shallow water, and along the borders of ditches (Peterson and McKenney, 1968). Smartweed is not controlled by Sonar® at the low concentrations used to treat Eurasian watermilfoil,

Spikerush (Eleocharis spp.)

Spikerushes (Family Cyperaceae) are annual or perennial sedges. Spikerushes are found in shallow water, marshes, and in wet soil, The culms of each plant are generally simple. The leaves are generally reduced to sheaths; very rarely the lowest leaf is bladebearing. Flowers are borne in spikes. There are approximately 120 species of spikerushes distributed in North America (Britton and Brown, 1970a). Some of the spikerush species indigenous to New York State include the creeping spikerush (Eleocharis fallax), blunt spikerush (Eleocharis obtusa), and dwarf spikerush (Eleocharis parvula). Engelmann spikerush (E. engelmannii) is listed as an endangered native plant species in NYS. Knotted spikerush (E. equisetoides), angled spikerush (E. quadrangulata) three-ribbed spikerush (E. tricostata), and long-tubercled spikerush (E. tuberculosa) are listed as threatened native plant species in NYS. Creeping spikerush, salt-marsh spikerush (E. halophila), and blunt spikerush are listed as rare native plant species in NYS. Spikerush is not controlled by Sonar® at the low concentrations used to treat Eurasisn watermilfoil.

3.0 ENVIRONMENTAL SETTING-SONAR

This section describes the environmental setting in which the proposed action, the use of the aquatic herbicide Sonar® is projected to occur. While this section presents the available data in as detailed an extent as is required, the information is generic for the State of New York.

3.1 GENERAL DESCRIPTION OF NEW YORK STATE AQUATIC ECOSYSTEMS

The aquatic ecosystems of New York State generally fall into four basic categories. These include standing freshwater systems (lakes, ponds, and reservoirs), flowing freshwater systems (rivers and streams), brackish systems (tidal estuaries), and saline coastal systems.

It is calculated that New York State has over 3.5 million acres covered by some type of surface water system (NYSDEC, 1967). That includes over 7500 lakes (NYSDEC, 1987), of which over 1500 are found in the Adirondack Mountains (NYSDEC, 1967). The Adirondack Mountains also contain over 16,700 miles of significant fishing streams. The state's largest lakes are Lake George, Lake Chautauqua, Oneida Lake, and the major Finger Lakes; Canadaigua, Keuka, Seneca, Cayuga, and Skaneateles (NYSDEC, 1967).

The specific characteristics of each aquatic system are partially determined by its physiographic setting within the state. Changes in the characteristics of each aquatic system will lead to changes in the endemic biota associated with that waterbody. Generally, waterbodies within New York State can be defined geographically by region and drainage basin location. Aquatic ecosystems in the eastern region, which includes the St. Lawrence/Lake Champlain/Black River basin, the Hudson-Mohawk basin, the Delaware basin, and Long Island are defined by either the Adirondack/Catskill mountain areas to the north or the New York Bight tidal estuarine area to the south. Aquatic ecosystems in the central region, which includes the Oswego-Ontario basin and the Susquehanna, are defined by areas of low relief with large areas of marshes to the north and broad, steeply sided valleys with limited natural storage capacity in the south. Aquatic ecosystems in the western region, which includes the Lake Ontario basin, the Erie-Niagara basin, the Genesee basin, and the Allegheny basin, are defined by the glaciated geology of that region (NYSDEC, 1967).

Waters in each of these basins are influenced by the composition of the geological formations found within the region. For example, waters in the Adirondack mountains and the Catskill mountains can be influenced by formations with little buffering capacity. In some lakes, this results in waters with pH values of less than 5 (NYSDEC, 1981b; ALSC, 1989). Surface water systems in the Erie-Niagara basin in western New York State are characterized by high levels of dissolved solids (140 to 240 ppm) and hard water (100 to 200 ppm, expressed as $CaCO_3$). Surface water in the Delaware River basin are characterized by low dissolved solid levels (averaging 37 ppm) and an average hardness of approximately 37 ppm. The dominant ions are silica, calcium, bicarbonate and sulfate (Archer and Shaughnessy, 1963). The dissolved solid concentrations in surface waters in the Champlain-Upper Hudson basin rarely exceed 500 ppm

(Giese and Hobba, 1970). In surface waters of the Western Oswego River basin, dissolved solid concentrations range from 50 to 300 ppm (Crain, 1975).

3.2 GENERAL CHARACTERIZATION OF AQUATIC PLANT COMMUNITIES IN NEW YORK STATE WATERBODIES

Aquatic plants are often the dominant biotic factors in pond settings and are important ecological features of larger waterbodies such as lakes and reservoirs.

The characteristics of plant communities in aquatic settings are determined by the type of waterbody in which the community is located. New York State, with its over 7500 lakes, contains an extensive array of freshwater systems. This diversity is further increased by the inclusion of streams, rivers, and other bodies of flowing water. Waterbodies vary in terms of color, pH, temperature, silt loading, bottom substrate, depth, rate of flow if it is a moving body, and watershed area. Each of these characteristics will affect, to some extent, the type and distribution of the plant communities in that waterbody. NYSDEC (1990) notes that the bottom morphology (shape) of a lake is a key factor is determining the type and extent of plant communities that are present. The chemical quality of the water is another factor that influences the distribution of plant species within a waterbody. Soft water lakes with a total alkalinity of up to 40 ppm and a pH of between 6.8 and 7.4 will often have sparse amounts of vegetation. Hard water lakes with a total alkalinity from 40 ppm to 200 ppm and a pH between 8.0 and 88 will have dense growths of emergent species that can extend into deeper water (Fairbrothers and Moul, 1965). Sculthorpe (1967) notes that the distribution of species within a waterbody is determined by the bottom substrate, light intensity (which is a function of depth and water clarity), and turbulence (currents or wave action).

Freshwater ecosystems include lentic ecosystems represented by standing waterbodies, such as lakes and ponds, and lotic ecosystems, which are represented by running water habitats. Lentic systems can be further subdivided in littoral, profundal, and benthic zones. The littoral zone is that portion of the waterbody in which the sunlight reaches to the bottom. This area is occupied by vascular, rooted plant communities. Beyond the littoral zone is the open water area, or limnetic zone, which extends to the depth of light penetration. This point of light penetration is called the compensation depth. This is the depth where approximately 1 % of the light incident on the water surface still remains. As a result of this decreased light, photosynthesis does not balance respiration in plants. Therefore, the light is not sufficient to support plant life. The strata below the compensation depth is called the profundal zone, The bottom of the waterbody, which is common to both the littoral zone and the profundal zone, is the benthic zone (Smith, 1980).

Lentic systems can be categorized based on ecological successional characteristics of the waterbody (Smith, 1980; NYSDEC, 1990; and Pullman, 1992). Succession is the ecological process by which one community is gradually replaced by a series of communities; tending to progress to a terminal community. In aquatic settings, the initial stage of succession is characterized by a lack of biota. Over a period of time, pioneering species colonize the waterbody, As the water and bottom substrates change as a result of movement of organic and inorganic sediments and nutrients into the waterbody, the organisms present change from those intolerant of higher organic material levels, to species that are more tolerant of the changes. Eventually, the waterbody can shift from a deep, sterile pool, to a shallow temporary pond, to an emergent marsh to eventually a terrestrial meadow. For additional information on lentic systems typical of NYS lakes, see Diet For a Small Lake (NYSDEC, 1990).

In lotic systems the distribution of plant communities is dictated by the velocity of the water flow and the nature of the bottom substrate. In fast moving waters, the system is usually divided into riffle and pool habitats. Riffles, which are areas of fast water, are centers of high biological productivity. However, the speed at which the water flows in these areas usually will not allow for rooted macrophytes to become established. Rooted vascular plant are more characteristic of pool habitats, which are interspersed with the riffle zones. In pools, the softer bottom substrate and the slower current velocities allow for the establishment of rooted plants. This is also the case for slower moving streams and rivers. In larger rivers, as with lakes, ponds, and reservoirs, depth becomes a determining factor for the distribution of plant communities (Smith, 1980).

Functionally, aquatic plants play important roles in the aquatic ecosystem. Aquatic macrophytes provide food and shelter for both vertebrate and invertebrate organisms and as spawning habitat for fish (Keast, 1984; (Gotceitas and Colgan, 1987; Schramm and Jirka, 1989; Hacker and Steneck, 1990: and Kershner and Lodge, 1990). The ability of the. nornrnunity to fill these functions, its value per se, is often a function of the species, density, and distribution of the members of that plant community. Daubenmire (1968) notes that plants in the genera Potamogeton and Scirpus are a favored food source for North American waterfowl, whereas muskrats (Ondatra zibethica) favor plants in the genera Carex, Sagittaria, and Typha. Brown et al. (1988) reported that vertically heterogeneous stands of aquatic macrophytes tended to contain more invertebrates than a community dominated by a single taxon. Therefore, opportunistic, rapid-growing species such as Eurasian watermilfoil, purple loosestrife, phragmites, and cattails, which develop dense monotypic stands in mature communities, would not he expected to offer the quality or diversity of habitat in such circumstances as more diverse communities would. Dionne and Felt (1991) note that high plant densities can interfere with the foraging ability and efficiency of piscivorous and insectivorous fish. Dense plant stands can directly or indirectly disrupt the utilization of macrophyte beds by fish and macroinvertebrates by affecting light penetration, temperature regimes, and water chemistry (Lillie and Budd, 1992).

Aquatic vegetation performs four basic functions in waterbodies (Fairbrothers and Moul, l965). These functions include: 1) modification of the dissolved gas content of the surrounding water; 2) provision of nutrient material suitable for food and the introduction of inorganic nutrients into the food cycle; 3) modification of the physical environment; and 4) the protection and provision of habitat for other organisms. In general, aquatic plants fulfill the preceding functions in the aquatic ecosystem. However, the extent to which those functions are fulfilled will depend on the location of the plant community (i.e. emergent community versus a deepwater community). The following sections more specifically address the type of plant community most likely to be involved in the use of Sonar® in New York State waterbodies. Furthermore, the roles that the individual species may play in that community are also described.

3.2.1 **Submerged, Deepwater and Floating Plant Communities**

Submerged plants are generally relegated to the littoral zone and include such genera as Potamogeton and Myriophyllum. Many of these macrophytes are rooted plants which complete the majority of their life cycle below the water surface, with only the reproductive structures extending above the water surface. Exceptions to this include plants in the genera Ceratophyllum and Utricularia. These plants do not have true roots, but are considered to be submerged plants found in the littoral zone (NYSDEC, 1990). Lemna and other free-floating species are generally found over the littoral zone and deeper water.

In ponded waters, generally a greater variety of plant genera is available to fulfill the necessary functions provided by the plant communities (Daubenmire, 1968). This occurs because of the small size of the ponds, which results in a reduction in the influence of wave action. Plant communities and large lakes can be influenced by wind driven waves which will restrict the distribution of plants in exposed areas. The functions described by Daubenmire include habitat for fish and invertebrates, food for waterfowl, and nesting or hiding areas for fish and other vertebrates, such as amphibians. Plants in the genera Ceratophyllum, Chara, Elodea, Naj as, and Potamogeton are the most common native species to fulfill these functions. These macrophyte species are generally the first macrophytes to advance over the bottom and will usually dominate the plant community which occupies that portion of the littoral zone at the pond margin to a depth of 7 meters.

In ponds, Daubenmire (1968) reports that floating plants, such as Lemna, are not affected by the depth of water with regards to distribution. The surface of a pond is a homogenous habitat for these plants, which will occur uniformly. Floating plants can be pushed by the wind from one area to another. Floating-leaved hydrophytes are common in shallow water habitats. These plants, such as the species Brasenia schreberi, Nuphar lutea and Nymphaea odorata, are limited to shallow water because they must produce a petiole of sufficient length to connect the root stock to the floating leaf,

Aquatic plant communities are commonly arranged by species along depth contours. These communities are comprised of either heterogeneous mixtures of species, or as is sometimes the case, they are comprised of monotypic stands of a single opportunistic macrophyte. The species diversity or richness of a plant community depends or sediment type, disturbance, and vegetation management efforts. The characteristics of the communities will, change with increasing depth as more shade tolerant species become dominant. Mosses, charophytes, several vascular species, and blue-green algae (Cyanobacteria) are the common constituents of the near-profundal zone. Open architecture species such as members of the genera Potamogeton are found in shallower, better lighted zones. Emergent species will typically dominate the shailowest water, but are usually accompanied by other vascular species.

Aquatic plants serve as food sources for a variety of organisms, including fish, waterfowl, turtles (snapping, Chelydra serpentina and painted, Chrysemys picta), and moose (Alces alces). Herbivores will consume fruits, tubers, leaves, winter buds and occasionally, the whole plant. Many species in the genera Potamogeton and Najas are considered to be valuable sources of food items. Plants in the genera Myriophyllum, Nymphaea, and Ceratophyllum are considered to be poor sources of food items (Fairbrothers and Moul, 1965). Nichols and Shaw (1986) note that Eurasian watermilfoil (Myriophyllum spicatum) is a poor source of food for waterfowl.

Submerged plants play an important role in supporting fish populations. Submerged plants provide food and shelter for rich and their young. Submerged plants serve as the substrate for the invertebrates that support fish populations. Smith et al. (1991) state that the production of forage fish and invertebrates generally increases in proportion to the submersed plant biomass. However, they conclude that populations of piscivorous fish tend to peak in water with intermediate levels of plant biomass, This is a function of the ability of the piscivorous fish, such as largemouth bass (Micropterus salmoides) to see their prey.

Submerged macrophyte stems and leaves may act as a substrate for a variety of microscopic organisms, called aufwuchs. Aufwuchs include bacteria, fungi, diatoms, protozoans, thread worms, rotifers and small invertebrates. The architecture of a particular plant species will also determine its suitability as a place for egg deposition for fish and amphibians. Additionally, the young of many fish species and some tadpoles will seek shelter in plant structures to evade predators.

Pullman (1992) notes that the architectural attributes of a particular plant species are a critical feature in the ability of that plant to function in support of fish populations. Those vertical plants with open architecture (some Potamogetons, Elodea, Cabomba, and a native species of Myriophyllum) provide more suitable habitat for fish than those plant species that form dense vertical mats or mats at the surface such as are formed by (Myriophyllum spicatum), and some Potamogetons (including Potamogetons crispus). Matted Eurasian watermilfoil plants have few leaves along their stems. The leaves are shaded and replaced by a dense leaf cover at the water's surface. The collection of vertical stems has limited habitat value.

Madsen et al (1991) supports this by noting that most native species are recumbent or have short stems and do not approach the water surface and therefore tend to support greater fish populations than mat forming macrophyte species. Variable height and leaf architecture will yield more diverse habitats.

Pullman (1992) concludes that, in general, most native aquatic plant species do not reach nuisance levels. It is generally the exotic, introduced species that reach nuisance proportions based on numbers or biomass and are considered to be weeds.

3.3 DISTRIBUTION AND ECOLOGY OF PRIMARY POTENTIAL AQUATIC MACROPHYTE TARGET SPECIES

As mentioned in Section 2.0, the proposed action is the use of the aquatic herbicide Sonar® for the control of nuisance aquatic vegetation located in the State of New York. NYSDEC (1981) defines nuisance vegetation as overabundant vegetation that may be aesthetically unpleasing, may interfere with effective and proper harvest of fishery resources, and may interfere with other recreational activities. Pieterse (1990) defines nuisance aquatic vegetation as an aquatic weed or "an aquatic plant which, when growing in abundance, is not desired by the manager of it place of occurrence". In some circumstances, the aquatic species of concern is an exotic or introduced species. Such a species is not indigenous to the area and was introduced either accidentally or on purpose. This is not to say that exotic aquatic macrophytes do not, in some circumstances, fulfill all of the benefits and functions of native species. This is discussed more thoroughly in Section 9.0. A plant species, whether native or exotic, becomes a nuisance when the population reaches some level of overabundance such that a problem with the waterbody is evident. However, because an aquatic species is an exotic or introduced species, it generally has the potential for a more rapid population growth for the following reasons.

Suter (1993) maintains that many of the severe man-caused effects brought upon natural biotic systems are caused by the introduction of exotic species. Introduced species are generally opportunistic in nature and are usually able to out-compete native species. Thus, they have can significantly alter the character of native plant communities or the ecosystems. Exotic species are considered pioneer species. Pioneer species are those organisms that possess a reproductive strategy that emphasizes efficient dispersal of propagules, rapid spread and growth rate, and sometimes high rates of biomass production emphasized by high productivity and rapid growth. These plants are able to occupy a wide diversity of habitats (Smith, 1980).

Invasive, exotic species have successfully extended their distribution through both natural and anthropogenic means on a world-wide basis. Nichols and Shaw (1986) and Wade (1990) note that an invasive aquatic macrophyte has the potential to infest a waterbody, then spread to the maximum extent of the available habitat. Following the initial invasion period, the production of the invasive species can attain a degree of stability and habitat equilibrium. Subsequently, the population of the invasive will fluctuate in response to the temporal and spatial dynamics of the aquatic environment (Nichols and Shaw, 1986; Wade. 1990). Usually, the equilibrium condition for the production of species such as Eurasian watermilfoil and curlyleaf pondweed is considered to be deleterious for most recreational and utilitarian uses as well as a disruptive influence on the production of native plants and animals.

Some exotic species do serve as target species for Sonar®. This is particularly true of Eurasian watermilfoil, curlyleaf pondweed and cabomba (See Section 2.3). However, other exotic species which have substantial populations in NYS are not considered to be target species. That includes waterchestnut (Trapa natans). The following sections describe the general distribution and ecology of the primary target macrophyte for Sonar®".

3.3.1 Eurasian Watermilfoil (Myriophyllum spicatum L.)

Eurasian Watermilfoil is an introduced exotic that is thought to be native to Eurasia and North Africa (Couch and Nelson, 1985). It is currently believed to have been introduced into the Chesapeake Bay region in the mid-1940s. Since then, it has spread across the St. Lawrence system, the Great Lakes region, and into British Columbia and Washington State (Aiken et al., 1979). It is found throughout the Tennessee Valley system and from Florida to Texas (Giesy and Tessier, 1979). As

of 1992, COLAM (1992) reports that Eurasian watermilfoil had been identified in lakes in 35 of New York State's 62 counties. In its 1993 Annual Report on the Aquatic Plant Identification Program, the Rensselaer Fresh Water Institute notes that 38 counties
had documented populations of Eurasian watermilfoil in 1993 (Eichler and Bombard, 1994), VDEC (1993) reports that over 35 lakes in Vermont have been infested with Eurasian watermilfoil as of 1992. That is up from approximately 5 lakes in 1982. Pullman (1993) reports that Eurasian watermilfoil had been identified in lakes in all 83 counties in Michigan by 1978.

Eurasian watermilfoil is a tolerant species that has been shown to grow well in a variety of aquatic habitats. Couch and Nelson (1985) note that the plant will thrive in all types of nutrient conditions (oligotrophic to eutrophic), both hard and soft water and under both brackish and freshwater conditions. The plant appears to grow best in fine, nutrient-rich sediments that do not contain more than 20% organic matter and requires a minimum light intensity of 1% to 2% of the available light (Smith and Barko. 1990). Kimbel (1982) reports that the colonization success of Eurasian watermilfoil in terms of growth and mortality is best in late summer months in shallow water on rich organic sediments. Eurasian watermilfoil's maximum growth rate occurs at temperatures ranging from 30 to 35° C (Smith and Barko, 1990). The plant utilizes both sediments and the surrounding surface water as sources of nitrogen and phosphorus (Smith and Barko, 1990). Barko and Smart. (1980) indicate that uptake by the roots is the primary means of obtaining phosphorus.

Eurasian watermilfoil grows in waters at depths of 0 to 10 meters (typically between 1 to 5 meters in depth). Eurasian watermilfoil will commonly grow as an emergent in circumstances where the water level of the lake slowly recedes (Aiken et at., 1919). Smith and Barko (1990) suggest that light intensity determines much of the distribution and morphology of Eurasian watermilfoil. While it grows in waterbodies with wide ranges in water clarity, in turbid waters growth is generally concentrated in the shallow areas (Titus and Adams, 1979). In relatively clear waters, Eurasian watermilfoil grows at much deeper depths and may not reach the water surface.

Pearsall (1920) considers Eurasian watermilfoil to be a deep water plant species, which he defines as a plant growing at a depth where light intensity is less than 15% of full sunlight. The common growth pattern for Eurasian watermilfoil is for the plant to initially colonize deeper waters, where it will generate a large quantity of biomass which extends to the surface (Coffey and McNabb, 1974). As the Eurasian watermilfoil reaches toward the surface, the lower leaves of the plant will be shaded out and will slough off. This creates a dense organic bed beneath dense beds of Eurasian watermilfoil and is part of the process that recycles nutrients back into the water column. The leaves and stems of Eurasian watermilfoil will concentrate at the surface of the waterbody, forming a thick canopy or mat which extends into shallower waters when the plant reaches sufficient densities.

Madsen et al. (1991a), in work done in Lake George, New York, noted that growth characteristics are facilitated by a high photosynthetic rate and a high light compensation point. Because of its high photosynthetic rate and correspondingly increased metabolic activity and productivity, the plant is able to grow at a significantly higher rate than that exhibited by native species such as Potamogeton spp. and Elodea canadensis. Additionally, with its high light tolerance, Eurasian watermilfoil will tend to grow closer to the waters surface than the native species that occur in low to medium light intensity regions of the littoral zone. This pattern allows for successful replacement or disruption of native vegetative communities. Madsen et al. (1991b) reported that dense growth of Eurasian watermilfoil in a bay in Lake George had significantly reduced the number of native species present.

Eurasian watermilfoil will overwinter with much of its green biomass intact, Because of its adaption to grow at lower temperatures than many native aquatic species, Eurasian watermilfoil is capable of tremendous growth at the very beginning of the growing season. The early timing of growth, in conjunction with its great ability to produce large quantities of biomass, further gives Eurasian watermilfoil a competitive advantage over most native aquatic macrophytes (Pullman, 1992). Smith and Barko (1990) report that the characteristic annual pattern of growth is for the spring shoots to begin growing rapidly as soon as the water temperature approaches l5°C. Pullman (1993) notes that this growth generally occurs before most native aquatic macrophytes become active. However, Boylen and Sheldon (1976) state that some native aquatic macrophytes, including Potamogoton robbinsii and P. amplifolius, will remain metabolically active at temperatures as low as 2°C.

As the shoots grow, the lower leaves slough off as a result of shading. As the shoots approach the surface, they branch extensively and form the characteristic canopy (mat) discussed earlier in this section. Biomass peaks at flowering in early

July, and then declines. If the population flowers early, a second biomass peak and subsequent flowering may be attained. It is common for Eurasian watermilfoil to adopt a stoloniferous habit in the autumn, growing prostrate over the surface of the lake sediment. This may also assist Eurasian watermilfoil in the displacement of competing native species through the acquisition of space when most native species are dormant. Variations in this growth pattern can occur as a result of differences in climate, water clarity and rooting depth.

Dispersal of Eurasian watermilfoil is primarily through the spread of vegetative fragments. Seed production has been reported, but is considered a minor contributor to the plant spread (Hartleb et al., 1993). Pullman (1993) notes that there is much circumstantial evidence indicating that Eurasian watermilfoil does not form a viable seed bank in infested lakes. Eurasian watermilfoil has a tremendous capacity for the formation of vegetative fragments. A viable plant can regenerate from a single node carried on a fragment released in the water. Fragmentation can occur from boating or skiing impacts, as well as from mechanical harvesting operations. Additionally, Madsen et at. (1988a) reports that autofragmentation (self-fragmentation) is common after peak seasonal biomass is attained. Often fragments released through autofragmentation bear adventitious roots. Madsen et al. (l988a) also noted that fragments are very durable, and resistant to extensive environmental stress.

Pullman (1993) concluded that Eurasian watermilfoil is supportive of fish populations during its initial expansion stages in a waterbody. However, he goes on to note that once Eurasian watermilfoil begins to dominate the plant community and form its characteristic dense mats, the lack of plant species diversity and associated water quality impacts will reduce the quality of the habitat for fish. Nichols and Shaw (1986) reported that Eurasian watermilfoil provides beneficial cover for fish, unless the cover is so dense that stunting of fish growth from overcrowding results. Eurasian watermilfoil has been shown to provide a better habitat for fish (Kilgore et al., 1989) and invertebrates (Pardue and Webb, 1985) than open water. However, Dvorek and Best (1982) found that Eurasian watermilfoil had the poorest invertebrate fauna populations out of 8 aquatic macrophyte species that were examined. Keast (1984) noted that fish abundance was 3 to 4 times greater in mixed native plant communities than in a plant community dominated by Eurasian watermilfoil. Nichols and Shaw (1986) noted that Eurasian watermilfoil is poor food for muskrats and moose and fair food for ducks, which will eat its fruit.

Eurasian watermilfoil is an opportunistic species that is commonly found growing in areas that are not highly disturbed (Pullman, 1992). However, Pullman goes on to report that Eurasian watermilfoil appears to significantly increase in numbers and in biomass in areas of disturbance. This is reflective of the high productivity rate of the species and its resulting ability to outgrow native plant species. Eurasian watermilfoil is an aggressive colonizer and is able to displace native submergent plant species in as little as 2 to 3 years (Aiken et at, 1979). Nichols and Shaw (1986) summarized that Eurasian watermilfoil has various physiological adaptations that allow the plant to rapidly propagate by vegetative means, an opportunistic nature for absorbing nutrients, a life cycle that favors cool weather and mechanisms that enhance photosynthetic activity.

Once it has formed dense stands, Eurasian watermilfoil interferes with, or prevents, recreational activities in a lake. Pullman (1993) notes that mats may constitute a safety hazard because they are not penetrable by boats and may hide submerged objects that could be struck by moving boats. He also notes that people can be placed at risk if they swim in dense areas of Eurasian watermilfoil due to the potential for entanglement.

3.4 DISTRIBUTION AND ECOLOGY OF OTHER POTENTIAL AQUATIC MACROPHYTE TARGET SPECIES OF SONAR®

In addition to the primary potential aquatic macrophyte target species discussed in Section 3.3, Sonar® is intended for use to potentially control other aquatic macrophyte species. While the opportunistic ecological behavior of Eurasian watermilfoil will lead to extensive growth and large quantities of biomass, under certain conditions, the following species may also reach a nuisance level. They include both introduced and native species.

Table 3-1 discusses the submerged, floating-leaved and floating macrophyte species that are potential targets for control by Sonar®. The sources of information for Table 3-1 include NYSDEC (1990), Fairbrothers and Moul (1965), Magee (1981). Hotchkiss (1972), and Martin et al (1951). These species are found throughout New York State, though the actual presence and distribution in a waterbody are dependent on the physical characteristics of that waterbody.

TABLE 3-1

DISTRIBUTION AND ECOLOGY OF
POTENTIAL SUBMERGED, FLOATING-LEAVED AND FLOATING
TARGET MACROPHYTE SPECIES

American Lotus (Nelumbo letua)

Found in ponds and quiet streams; is at the northern edge of its geographic distribution in NYS

Bladderwort (Ultricularia spp.)

Found in ponds, lakes and sluggish streams throughout New York State (NYS); is considered of little food value to birds and mammals, but is a provider of cover for fish

Common Coontail (Ceratophyllum demersum)

Found in shallow ponds and slow streams throughout NYS; provides good shelter for young fish and supports insects that are eaten by fish; its fruits are eaten by waterfowl

Common Elodea (Elodea canadensis)

Found in ponds, lakes and sluggish streams throughout NYS; provides shelter for fish; used as food by waterfowl

Egeria, Brazilian Elodea (Egeria densa)

Found in ponds, lakes and sluggish streams; is a rare and exotic species in NYS; is considered to have escaped into the natural environment

Fanwort (Cabomba caroliniana)

Found in ponds and quiet streams in Southern regions of NYS; provides cover and food for fish; not an important food for waterfowl or mammals

Naiad (Najas spp.)

Grows in shallow ponds, lakes and sluggish streams throughout NYS; all parts of these plants are eaten by waterfowl

TABLE 3-1 (CONTINUED)
DISTRIBUTION AND ECOLOGY OF
POTENTIAL SUBMERGED, FLOATING-LEAVED AND FLOATING
TARGET MACROPHYTE SPECIES

Parrotfeather (Myriophvllum brasiliense)

Grows in shallow ponds, lakes and sluggish streams throughout most of NYS; poor food source; good shelter for invertebrates and fish

Pondweed (Potamogeton spp.)

Found in sluggish streams, lakes and ponds throughout NYS; all portions of the plant are eaten by birds and muskrats

Watermilfoil (Myriophyllum spp.)

Native watermilfoil species are found in ponds, lakes and sluggish streams throughout NYS; is considered a low-grade duck food; is considered to be good habitat and shelter for fish and macroinvertebrates

Spatterdock (Nuphar luteum)

Found in sluggish streams, ponds, small lakes and swamps throughout NYS; low wildlife food value

Waterhyacinth (Eichornia crassipps)

Rare and introduced in NYS; found in ponds, lakes and sluggish streams

Waterlily (Nymphaea spp.)

Found in shallow ponds, lakes and swamps throughout NYS; seed and rootstocks are eaten by ducks and marshbirds, beaver and moose eat the foliage, invertebrates utilize the undersides of leaves as shelter

Waterprimrose (Ludwigia spp., including waterpurslane (Ludwigia palustris)

Found in streams and springy areas throughout NYS; serves as a food source for birds and grazing mammals

TABLE 3-1 (CONTINUED)
DISTRIBUTION AND ECOLOGY OF
POTENTIAL SUBMERGED, FLOATNG-LEAVED AND FLOATING
TARGET MACROPHYTE SPECIES

Watershield (Brasenia schreberi)

Grows in ponds, lakes, and along margins of sluggish streams; plants provide shade and shelter for certain fish; fruits are eaten by various species of ducks

3.5 ROLE OF POTENTIAL AQUATIC MACROPHYTE TARGET SPECIES IN PLANT COMMUNITIES WITHIN NEW YORK STATE WATERBODIES

As discussed in Section 3.2, aquatic macrophytes fulfill valuable functions in the aquatic environment. They assist in oxygenation of the water, recycling of nutrients, and provide nesting and shelter areas for fish, amphibians, birds and mammals. Aquatic macrophytes serve in the stabilization of banks along watercourses and are a food source for a variety of organisms, including both invertebrates and vertebrates. The ability of a particular macrophyte to perform these functions and the quality of that function often depends on the characteristics of the entire aquatic community. Heterogeneous stands of plant species generally offer more of these functions than a monotypic stand (dominated by a single species). Heterogeneous stands have a greater vertical distribution of niches, which aquatic organisms that are dependent on the vegetation may fill. Additionally, the horizontal distribution of the aquatic plant communities will affect the functions and values that the individual species may offer. Patchy communities, with a variety of vegetative species spread over the available substrate, tend to offer a greater variety in habitats than a community dominated by a single species that completely covers the substrate. However, if that single species community is localized and is the only available habitat in a large aquatic setting, then at least some of the functions generally offered by aquatic vegetation would be offered. This circumstance may be evaluated in a lake management plan that would determine the goals and objectives of the vegetation management needs for that waterbody. Restoration of a mixed community of desirable plant species is likely to require initial removal of a monotypic plant stand.

3.5.1 Submerged, Floating-leaved, and Floating Plant Communities

Lilllie and Budd (1992) provide a definitive evaluation of the quality of habitat offered by Eurasian watermilfoil. In their study, conducted on a lake in Wisconsin, Lillie and Budd utilized an index of plant habitat quality and quantity to describe the following; 1) horizontal visibility within macrophyte beds; 2) the amount of shading afforded by the surface canopy; 3) the amount of available habitat for macroinvertebrate attachment: 4) the relative amount of protection afforded fish by the plants; and 5) the degree of crowding or compaction among plants. The results of their study indicated that the edges of Eurasian watermilfoil beds potentially provide more available habitat for macroinvertebrates and fish than interior portions. This conclusion was based on their observation that habitat space was more optimal at the edges, than in the center of the beds where stem crowding and self-defoliation resulted in a lack of vertical architecture due to the formation of surface mats. They noted that as Eurasian watermilfoil densities increase from sparse to dense, habitat value for prey species increased. However, as the vegetative density increased in Eurasian watermilfoil stands, a reduction in habitat for macroinvertebrates reduced the habitat quality for small fish. Habitat value for predator fish species initially increased as Eurasian milfoil first colonized areas, but, then decreased as plant crowding impacted the ability of the predators to access their prey.

The work by Lillie and Budd (1992) suggests that in relatively new or small Eurasian watermilfoil beds or in heterogenous communities where watermilfoil is a component, habitat functions and values of this plant are consistent with native plant species. However, it must be recognized that areas occupied by small, new or partial Eurasian watermilfoil stands may become dominated by this species within one or two seasons (Lillie and Budd, 1992).

In work conducted by Keast (1984) in a lake in Ontario, Canada, Eurasian watermilfoil significantly modified the habitat available to fish and macroinvertebrates. Keast noted that since the advent of Eurasian watermilfoil in his study area, significantly fewer bluegill (Lepomis macrochirus) were observed, but greater numbers of black crappie (Pomoxis nigromaculatus) and golden shiner (Notemigonus crysoleucus) were seen. He reported 3 to 4 times as many fish feeding in native plant beds as in the Eurasian watermilfoil beds.

The most critical impact Keast (1984) noted was to prey organisms. Keast reported that significantly fewer macroinvertebrates were seen in the watermilfoil beds than in a native plant community composed of Potamogeton and Vallisneria. He found 3 to 7 times greater abundance of 5 invertebrate taxa in the native plant communities and noted that foliage of the native plants supported twice as many invertebrates per square meter. Keast observed twice as many insect emergences in the native plant community as in the Eurasian watermilfoil beds.

Other recent studies have documented the impacts to the aquatic environment by the invasion of Eurasian watermilfoil. Madsen et al. (1991) noted a sharp decline in the number of native species per square meter in a bay in Lake George, New York. The decline was due to the suppression of native species by Eurasian watermilfoil. The decline was from 5.5 species per square meter to 2.2 species per square meter over a 2-year period.

Honnel et at. (1992) noted that in ponds containing Eurasian watermilfoil, dissolved oxygen levels were significantly lower than dissolved oxygen levels in ponds dominated by native plants. Additionally, they note that pH levels were higher in Eurasian watermilfoil than in native plant dominated ponds.

3.6 GENERAL CHARACTERIZATION OF AQUATIC VEGETATION MANAGEMENT OBJECTIVES FOR THE USE OF SONAR®

Aquatic vegetation management becomes necessary when the populations or biomass of aquatic macrophytes in a waterbody become so great that they impact some function or use of that waterbody. This is equally true for introduced exotic plant species, such as Eurasian watermilfoil, which displace native species that may possess greater ecological value. Those deleterious effects could include reduction in fish populations or quality of the fishery, angler success or waterfowl use, restrictions in boating or swimming, and clogging of intake pipes. Additionally, the scenic beauty on the lake and value of lakeside property will be significantly reduced as a result of the uncontrolled spread of an invasive species.

The primary management objective for the use of Sonar® is the control of overabundant submerged aquatic weeds, particularly Eurasian watermilfoil and curlyleaf pondweed. How Sonar® is to be used within the waterbody will depend on the aquatic plant management objectives for the individual waterbody. It is important that these objectives be identified by the lake association or organization governing the use of a waterbody. Factors which may need to be considered in developing the objectives include the size of the lake or waterbody and whether the waterbody is to be used for potable water, swimming, boating, and fish or waterfowl management. Improvement or maintenance of aesthetic, scenic, and property values may also require aquatic plant management. Additionally, information on the development of lake management objectives can be found in Chapter 5 of Diet For a Small Lake (NYSDSC, 1990).

4.0 GENERAL DESCRIPTION OF SONAR® AND ITS ACTIVE INGREDIENT FLURIDONE

4.1 GENERAL DESCRIPTION OF SONAR® A.S. AND SRP FORMULATIONS

Sonar® is a systemic aquatic herbicide used in the management of aquatic macrophytes in freshwater ponds, lakes, reservoirs, drainage canals, irrigation canals, and rivers. The active ingredient of Sonar® is fluridone. Two formulations of Sonar® are registered in New York State. Sonar® A.S. (Aqueous Suspension) is a liquid formulation containing 41.7% fluridone and 58.3% inert ingredients. Sonar® SRP (Slow Release Pellets) is a dry material containing 5.0% fluridone and 95.0% inert ingredients.

4.1.1 Active Ingredients

The active ingredient in Sonar® is fluridone (l-methyl-3-phenyl-5-[3-(trifluoromethyl)phenyl)- 4[1H]-pyridinone). Technical fluridone is an off-white to tan, odorless crystalline solid. It melts at between 151 to 154° C. The vapor pressure of fluridone is less than 1×10^{-7} mm Hg at 25°C. Fluridone is stable to hydrolysis at a pH of 3, 6 and 9. The partition coefficient (log K_{ow}) for fluridone in n-octanol/water is 1.87. Fluridone is not corrosive.

4.1.2 Inert Ingredients

The primary inert ingredient in Sonar® A.S. is water. Other inert ingredients are added to serve as wetters and dispersants in the formulation and to prevent freezing during storage. Sonar® A.S. and Sonar® SRP do not contain any inert ingredient listed on the USEPA List 1- Inerts of Potential Toxicological Concern or List 2 - Potentially Toxic Inerts/High Priority for Testing. The primary inert ingredient in Sonar® SRP is clay. Small amounts of a binder are added to maintain the integrity of the pelleted formulation.

4.1.3 Product Contaminants

There are no toxicological concerns associated with product impurities in Sonar® herbicides as formulated.

4.2 SELECTION OF SONAR® SRP VERSUS SONAR® A.S.

The selection of Sonar® SRP versus Sonar® A.S. should be based on the management objectives of the aquatic macrophyte control program for the particular waterbody. The permit restrictions for the products should also be considered, noting that Sonar® A.S. is only registered for the management of Eurasian watermilfoil. The selection of one formulation or the other is related to maintaining an appropriate concentration of fluridone for a sufficient amount of time to allow for uptake by the target macrophyte. Generally, Sonar® SRP is more appropriate for moving water because it releases fluridone over a longer period of time than the A.S. formulation. This will allow for a longer exposure time than the liquid formulation which would tend to be more rapidly diluted by untreated water.

Sonar SRP is most effective when applied while the target submerged plants are low growing in the water column and where bottom sediments are sands or other firm substrates. Sonar® A.S. is most effective where target submerged plants have grown to near the water surface. Sonar A.S. performs well when applied over soft muck or organic sediment.

4.3 DESCRIPTION OF USE

Sonar® is used as a systemic herbicide for the control of unwanted aquatic macrophytes in lakes, ponds, reservoirs, slow moving rivers, drainage canals, and irrigation canals. Sonar® A.S. can be applied through surface application, subsurface application, or by bottom application just above the hydrosoil. Sonar® SRP is applied through any type of broadcast applicator.

4.4 MODE OF ACTTON/EFFICACY

Sonar® is a systemic herbicide that is absorbed from the water column by plant shoots and from the hydrosoil by roots. The active ingredient in Sonar®, fluridone, inhibits the biosynthesis of carotenoid pigments within susceptible plants. Carotenoid pigments protects the photosynthetic pigment chlorophyll from photodegradation. Without the carotenoid pigments, chlorophyll is photodegraded and the plant is unable to carry on photosynthesis, Photosynthesis is required by the plant to produce carbohydrates necessary for metabolism (Elanco, 1981 and USEPA, 1986a). Specifically, the application of fluridone results in the accumulation of the colorless carotenes, phytoene and phytofluene, and lack of formation of the colored carotenoid, ß-carotene. In the absence of ß-carotene, chlorophyll is destroyed and the chloroplasts are disrupted in the sunlight causing cellular bleeding (Bartels and Watson, 1978 and Kowalczyk-Schröder and Sandmann, 1992).

Sonar®, and its active ingredient fluridone, have been shown to effectively control susceptible aquatic macrophytes. Eurasian watermilfoil and curlyleaf pondweed have been shown to be highly sensitive to fluridone. Pullman (1993) reported the removal of Eurasian watermilfoil and curlyleaf pondweed and the restoration of the native plant community following the treatment of a lake in Michigan with Sonar® at a rate of 13.6 ppb. Pullman (1993) cited more than two dozen other lake treatments in Michigan using application rates of between 8 ppb to 29 ppb to successfully control Eurasian watermilfoil and curlyleaf pondweed.

Sonar® is a slow-acting herbicide that requires an extended period of contact with the target macrophytes for the herbicidal effects to be induced. Netherland and Getsinger (1992) note that control of Eurasian watermilfoil with fluridone may take several weeks. DowElanco (1990) stated that it generally takes 30 to 90 days for Eurasian watermilfoil to drop out of the water column after treatment.

4.5 APPLICATION C0NSIDRATIONS THAT MAXIMIZE THE SELECTIVITY OF SONAR®

Application considerations should include those conditions described in 6 NYCRR Part 326. Under those considerations, fluridone may only be used as follows:

1. Application of aqueous suspension formulations are permitted in water of the State at application rates not to exceed 50 ppb of the active ingredient fluridone.

2. Application of pellet formulations are not permitted in waters less than two feet deep.

3. Swimming is not allowed in treated areas for a period of 24 hours following the application.

SONAR cannot be used with 1320 feet of any functioning potable water intake and users must comply with all other federal and state approved label requirements. Further, it must be noted that Sonar® A.S. is only permitted for the treatment of Eurasian watermilfoil. The following factors should be considered in the application of Sonar® to ensure maximum selectivity of the product.

4.5.1 Time of Application

It is recommended that Sonar® be applied as early in the growing season, as possible. Eurasian watermilfoil initiates productivity and metabolic activity at an earlier time than native plants (Smith and Barko, 1990). They report that the characteristic annual pattern of growth is for the spring shoots to begin growing rapidly as soon as the water temperature approaches 15° C. Pullman (1993) notes that this growth generally occurs before most native aquatic macrophytes become active. However, Boylen and Sheldon (1976) state that some native aquatic macrophytes, including Potamogeton robbinsii and P. amplifolius, will remain metabolically active at temperatures as low as 2° C. As a result of those growth characteristics, an early season application is recommended.

Utilizing an early growing season application would allow for the treatment of Eurasian watermilfoil while the remaining plant community is still dormant. Additionally, such applications would occur while the water is sufficiently cold to prevent recreational use (Pullman, 1994). Based on observations made in Michigan, Pullman (1993) noted that several broadleaf pondweeds may be moderately to highly susceptible to fluridone at application rates of 15 to 20 ppb, if the application occurs as these plants begin to grow. Though again, the spring growth of these species occurs after initiation of the growth of Eurasian watermilfoil.

4.5.2 Rate of Application

The registered application rates are described on the labels attached as Appendix A. Application rates for individual treatments may be varied to reflect the potential for water exchange in the treated area and for the susceptibility of target plants. Where treatments are being applied on a whole lake basis, with minimal opportunity for dilution by untreated water, application of Sonar® A.S. at low fluridone concentrations of 10 to 12 ppb has provided control oe Eurasian watermilfoil. Higher rates may be required where applications are made to portions of a water body and where water movement will cause dilution with untreated water. Such conditions would be based on the characteristics of an individual site.

It is the objective of this GEIS, under the SEQR process, to objectively present all pertinent facts associated with the potential use of these products as currently registered in the State of New York. The information that has been presented in the GEIS is a compilation of facts that have been shown in various studies. While it is true that lower applications rates may be efficacious, this is usually in entire waterbodies where the concentrations can be maintained for a sufficient period of time, In larger waterbodies where partial area control may be attempted, a higher concentration (but not exceeding the registered application concentration) would be required to compensate for dilution from untreated waters. It is for this reason that the NYS registered labels for Sonar® SEP and A.S. give the user a range of application rates such that a variety of site circumstances can be addressed.

4.5.3 Method of Application

The method of application should be chosen based on the formulation of Sonar® to he used, which is a function of the management objectives of the control program. Sonar® A.S. can be applied through surface application, subsurface application, or by bottom application just above the hydrosoil, if plant development permits. Sonar® SRP is applied through any type of broadcast applicator. Sonar® should be applied as evenly as possible over nuisance plant zones. However, certain lake basin morphometries may require that the material be applied uniformly over the entire lake. This should be done to enhance the selectivity of the Sonar® application.

4.5.4 Species Susceptibility

The potential target macrophytes discussed in Section 2.0 are susceptible to Sonar®. Susceptibility is related to the concentration of Sonar® applied to the system. Table 4-1 lists the species considered to be susceptible to Sonar®.

4.5.5 Dilution Effects

As previously noted, the important factor regarding the efficacy of Sonar® is the ability to keep a sufficient concentration of fluridone in contact with the plant for a sufficient time to allow for uptake by the target macrophyte. To prevent the dilution of the herbicide from reducing efficacy, several recommendations may be made. Ponds should be treated at one time. If lakes or reservoirs are being treated, it is recommended that treated areas be greater than 5 acres. To obtain effective plant control, spot treatments should not be applied to small (less than 5 acre) areas in large water bodies, such as when narrow boat lanes or dock areas are being treated. Application periods should be chosen when heavy rainfall is not expected. Where possible, the efficacy may be improved by restricting the flow of water. Whole lake applications provide the greatest opportunity for the long-term restoration of native plant communities.

TABLE 4-1.

SPECIES CONSIDERED SUSCEPTIBLE TO SONAR

American Lotus (Nelumbo lutea)
Bladderwort (Ultricularia spp.)
Common Coontail (Ceratophyllum demersum)
Common Elodea (Elodea canadensis)
Egeria, Brazilian Elodea (Egeria densa)
Fanwort (Cabomba caroliniana)
Naiad (Najas spp.)
Parrotfeather (Myriophyllum brasiliense)
Pondweed (Potamogeton spp.)
Watermilfoil (Myriophyllum spp., including Eurasian watermilfoil, M. spicatum)
Spatterdock (Nuphar luteum)
Waterhyacinth (Eichornia crassipes)
Waterlily (Nymphaea spp.)
Waterprimrose (Ludwigia spp., including waterpurslane (Ludwigia palustris)
Watershield (Brasenia schreberi)

Sources: Payne, 1992, Pullman, 1993 and
the **NYS** approvcd labels for Sonar 81W and Soner0 A-S.

4.6 FLURIDONE PRODUCT SOLTUBILITY

Fluridone is slightly soluble in organic solvents such as methanol, diethyl ether, ethylacetate, chloroform, and hexane. Fluridone has a water solubility or 12 ppm, which is considered to be medium solubility. The solubility of fluridone in water is greater than the 0.05 ppm use rate on the NYS SLN label for Sonar® A.S.

4.7 SURFACTANTS

Surfactants are not used with Sonar® products when used as labeled in New York.

4.8 FATE OF FLURUDONE AND ITS PRIMARY METABOLITE IN THE AQUATIC ENVIRONMENT

Various studies have indicated that photolysis is the primary degradation mechanism for fiuridone in aquatic ecosystems (Saunders and Mosier, 1983 and Muir and Grift, 1982). Microbial degradation of fluridone is documented to occur in laboratories (Mossler et al., 1991); however, photolysis generally occurs much more quickly (Muir and Grift, 1982), West and Parka (1981) also reported that the photolytic action occurs rapidly and is not influenced by the type of dispersal mechanism used to introduce Sonar®. Variables which may affect the rate of photolysis are those variables associated with sunlight penetration of the water column and sunlight intensity. They include geographic location, date of application, water depth, turbidity, weather, and weed cover (West et al., 1983).

West et al. (1983) identified 1-methyl-3-(4-hydroxyphenyl)-5-[3-(trifluoromethyl)phenyl]-4[1H]-pyridinone as the primary metabolite in fish. The same metabolite was identified as a minor metabolite in water and hydrosoil by Muir and Grift (1982). West et al. (1983) also identified 1 ,4-dihydro-1-methyl-4-oxo-5-[3-(trifluoromethyl)phenyl]-3-pyridinone as the major hydrosoil metabolite in hydrosoil studies conducted in laboratory settings. They note that the laboratory hydrosoil metabolite has not been identified in the hydrosoil of small ponds under natural conditions. Saunders and Mosier (1983) identified benzaldehyde, 3-(trifluoromethyl)-benzaldehyde, benzoic acid, and 3-(trifluoromethyl)-benzoic acid as photolytic breakdown products of fluridone added to a methanol/water solution in the laboratory.

Saunders and Mosier (1983) also identified N-methylformamide (NMF) as a photolytic breakdown product of fluridone which was added to a methanol/water solution in the laboratory. NMF has been shown to be teratogenic in rabbits at high doses and can penetrate human skin (Gaines, 1989). Early investigators were concerned with the possibility of NMF being produced by the breakdown of fluridone in the natural environment. However, NMF has never been identified under natural conditions (Gaines, 1989 and Osborne et al., 1989). Dechoretz (1991) did not identify NMF in water samples collected from ponds in California treated with aqueous suspension and pelleted formulations of Sonar®. West et al. (1990) did not identify NMF in water or hydrosoil samples collected from two ponds in Florida treated with Sonar® A.S. and Sonar® SRP at application rates of 0.15 ppm. In three ponds in Massachusetts, Smith et al. (1991) applied Sonar® A.S. and Sonar® SRP at a concentration rate of 0.15 ppm. Analysis of water samples collected from the ponds did not detect for NMF. Osborne et al., did not find NMF in water samples from ponds treated with up to 446 ppb fluridone.

4.8.1 Water (Aerobic and Anaerobic)

USEPA (1986a) reports that, under anaerobic conditions, fluridone has a half-life of 9 months and under aerobic conditions has an average half-life of 20 days. In field trials in ponds and lakes, using pelleted and aqueous Sonar® formulations, West et al. (1983) reported that the average maximum concentration for fluridone occurred 1 day after treatment in ponds (0.0871 ppm) and lakes (0.026 ppm). Observed concentrations are, of course, dependent on use rate.

Ponds, which were 1.2 hectares and smaller, were located throughout the U.S., including New York State. Treatment in this study was on a whole pond basis. Lakes were larger than 1.2 hectares and were located in Florida and Panama. Areas of 0.8 to 4.0 hectares were treated in lakes, West et al. (1983) reported the maximum average concentrations of fluridone in water after treatment using a pelleted formulation of Sonar® (Sonar® 5P), occurred 2 weeks after treatment in ponds (0.025 ppm) and 1 day after treatment in lakes (0.022 ppm). The delay in reaching the maximum concentration in the pelleted formulation is due to the time involved in the breakdown of the clay pellet and the subsequent release of fluridone. West et al. (1983) noted that the average fluridone concentrations in the water from the pelleted formulation were similar or less than the average fluridone concentrations in the water from the aqueous formulation. Additionally, their results indicated that, once the maximum fluridone concentrations were reached, the dissipation rates between the two formulations were similar.

Langeland and Warner (1986) supported the work conducted by West et al. (1983). In the study conducted by Langeland and Warner, two ponds In North Carolina were treated with 2.27 kg ai/ha and 1.14 kg ai/ha of Sonar® A.S. respectively. One additional pond in Virginia was treated with Sonar® 5P, a pelleted formulation, similar to Sonar® SRP. Their results indicated that between 64 and 69 days were required to reach no detectable levels of fluridone in the Sonar® A.S. treated ponds. In the Sonar® 5P treated lake, the maximum fluridone concentration (44.4 ppb) was reached 17 days after treatment, reflecting a time lag necessary for the fluridone to dissociate from the pellet formulation. Concentrations then decreased until 51 days after treatment, when a small increase in the fluridone concentration (from 20.9 to 28.9 ppb) in water was observed. Langeland and Warner speculated that this was the result of the release of fluridone back into the water from stressed vegetation.

West et al. (1983) reported that the half-life for fluridone in pond water treated with Sonar® A.S. ranged from *5* to 60 days. They were unable to calculate a half-life figure for the pelleted formulation of Sonar®. This was because fluridone was degrading at the same time it was being released from the pelleted formulation, resulting in a steady state concentration. Muir et al. (1980) reported a half-life for fluridone in water at a treatment level of 0.70 ppm of 4 days.

4.8.2 Sediment

Fluridone will adhere to sediment particles and organic material within the sediment. Elanco (1981) reported that fluridone will gradually desorb from the hydrosoil into the water column where it will photodegrade. Malik and Drennan (1990) noted that ph can be a controlling factor in adsorption, with the strength of adsorption increasing with lower pH levels. USEPA (1986a) notes that the half-life of fluridone in the hydrosoil is 90 days. West et al. (1979) reported a sediment maximum residue concentration equivalent to 16% of the fluridone theoretically applied to a pond in New York State. The application rate was 2.7 kg/ha of an aqueous fluridone formulation. That residue concentration decreased to 3% of the applied amount after 112 days. West et al. (1983) calculated a half-life of 3 months for fluridone in the hydrosoil of ponds. Additionally, they noted in 20 field trials that the laboratory hydrosoil metabolite does not form under natural conditions. West et al. (1983) also reported that studies on sediment-water systems indicated that fluridone tends to establish an equilibrium concentration between the water and sediment. Removal of fluridone from the water through photolysis results in the desorption of fluridone from the sediment into the water column to maintain the equilibrium.

4.8.3 Plants

Muir et al. (1980), using exaggerated application rates, reported a maximum residue concentration of 63.71 ppb of fluridone in duckweed (Lemna minor) following exposure to 5.0 ppm of fluridone in water. West et al. (1979) reported a maximum fluridone residue concentration of 3.98 ppm in Elodea canadensis, 7 days after treatment with an aqueous solution of fluridone that resulted in a water column concentration of 0.30 ppm at the time of application.

There is no information available on studies of herbivorous animals that consume aquatic vegetation containing fluridone residues. However, based on the low bioaccumulation rates reported in plants and the high levels of fluridone necessary to produce a toxic response in mammals and birds, it is not expected that herbivorous animals would be impacted by the use of fluridone at the registered application rates.

4.8.4 Fish

Based on all available fluridone residue data, USEPA has established a tolerance level of 0.5 ppm as adequate to protect human health from consumption of fish and crayfish (40 CFR and 180.420). The tolerance expressions assume an application at the maximum rates listed on the Federal Sona® labels. West et al. (1983) reported that the maximum residue in the edible tissue of fish (the filet) occurred 1 day after treatment using Sonar® A.S. (reported 0.132 ppm), 14 days after treatment (reported 0.528 ppm) in inedible tissue (the viscera) and 14 days after treatment in whole fish (reported 0.399 ppm). They also reported a maximum residue level in the edible tissue of fish occurred 1 day after trearment using a pelleted formulation of Sonar® (reported 0.067 ppm), 28 days after treatment in inedible tissue (reported 0.268 ppm) and 28 days after treatment in whole fish (reported 0.185 ppm).

Muir et al. (1980) observed a maximum concentration of 0.17 ppb of fluridone in fathead minnows (Pinephales promelas) following exposure to 0.070 ppb of fluridone in water. Additionally, they noted that the maximum concentration was detected 9.6 days after treatment. In ponds treated at an application rate of 0.1 ppm, Arnold (1979) noted fluridone concentration residues of 0.054 ppm in green sunfish (Lepomis cyanellus) one day after application; concentration residues in pumpkinseed sunfish (Lepomis gibbosus) of 0.023 ppm and in largemouth bass (Micropterus salmoides) of 0.010 ppm 7 days after application; concentration residues in black bullhead (Ictalurus melas) of 0.010 ppm 14 days after application; and no detectable concentration residues in pumpkinseed sunfish and largemouth bass after 27 days after application.

The consensus of the scientific literature is that fluridone concentrations in fish generally reflect the concentrations in water. As the residues are removed from the water column, they clear from fish tissues. In their work, West et al. (1983) observed that concentrations of fluridone in fish were at non-detectable levels following dissipation of the material from the water column. This supported the observations made by Muir et a1. (1980).

There is no information available on studies of fish-eating mammals or birds that consume fish containing fluridone residues. However, based on low bioaccumulation rates reported in fish and the high levels of fluridone necessary to produce a toxic response in mammals and birds, it is not expected that piscivorous animals would be impacted by the use of fluridone at the registered application rates.

4.8.5 Mammals

Absorption/excretion studies in rats indicate that a single oral dose of fluridone is rapidly absorbed and extensively metabolized and primarily excreted in the feces. Arnold (1979) noted that the fluridone dose was excreted within 72 hours. More than 80% was excreted in the feces and a trace was excreted in the urine.

4.8.6 Bioaccumulation/Biomagnification

USEPA (1986a) states that fluridone has a low potential for accumulation in fish. West et al, (1983) identified a total average bioconcentration factors for total fluridone residues of 1.33 for edible tissue, 7.38 for inedible tissue, and 6.08 for whole body. These data were obtained from 175 fish samples collected from across the country, including New York State. Muir et al. (1980) reported bioconcentration factors of up to 85 in duckweed following exposure to 5-0 ppm of fluridone in water. West et al. (1979) reported bioconcentration factors ranging from 0 to 15.5 in vascular plants following exposure to 0.10 ppm of fluridone in water. These peak values of fluridone residues were followed by a decline in concentrations as fluridone dissipated from the water column. No circumstance was identified in the scientific literature where fluridone irreversibly accumulated in biological tissues and remained after the dissipation of fluridone from the water column.

4.9 FLURIDONE RESIDUE TOLERANCES

The following residue tolerances have been established in accordance with applicable federal regulations.

4.9.1 Water

The USEPA designated an acceptable residue level for fluridone in potable water of 0.15 ppm. This concentration is based on the maximum application rate for fluridone as registered under FIFRA (USEPA, 1986a). NYS DOH has established an acceptable level of 0.05 ppm for unspecified organic compounds in drinking water that applies to fluridone residues.

4.9.2 Fish/Shellfish

The USEPA has designated a tolerance of 0.5 ppm for residues of fluridone and its primary rnetabolite (metabolite II) in fish (USEPA, 1986) and crayfish (40 CFR §180.420).

4.9.3 Crops/Agricultural Products

USEPA (1986) and 40 CER § 180.420 have designated the following residue tolerances for crops irrigated with water containing fluridone residue concentrations of 0.15 ppm:

Commodities	Parts per Million
Avocados	0.10
Citrus	0.10
Cottonseed	0.10
Cucurbits	0.10
Forage grasses	0.15
Forage legumes	0.15
Fruiting vegetables	0.10
Grain crops	0.10
Hops	0.10
Leafy vegetables	0.10
Nuts	0.10
Pome fruit	0.10
Root crops, vegetables	0.10
Seed and pod vegetables	0.10

Small fruit	0.10
Stone fruit	0.10

Additionally, residue tolerances have been established for the following raw agricultural commodities by USEPA (1986a) and 40 CFR § 180.420:

Commodities	Parts per Million
Cattle, fat	0.05
Cattle, kidney	0.10
Cattle, liver	0.10
Cattle, meat (except liver and kidney)	0.05
Cattle, mbyp	0.05
Eggs	0.05
Goats, fat	0.05
Goats, kidney	0.10
Goats, liver	0.10
Goats, meat (except liver and kidney)	0.05
Goats, mbyp	0.05
Hogs, fat	0.05
Hogs, kidney	0.10
Hogs, liver	0.10
Hogs, meat (except liver and kidney)	0.05
Hogs, mbyp	0.05
Horses, fat	0.05
Horses, kidney	0.10
Horses, liver	0.10
Horses, meat (except liver and kidney)	0.05
Horses, mbyp	0.05
Milk	0.05
Poultry, fat	0.05
Poultry, kidney	0.10
Poultry, liver	0.10
Poultry, meat (except liver and kidney)	0.05
Poultry, mbyp	0.05
Sheep, fat	0.05
Sheep, kidney	0.10
Sheep, liver	0.10
Sheep, meat (except liver and kidney)	0.05
Sheep, mbyp	0.05

5.0 SIGNIFICANT ENVIRONMENTAL IMPACTS ASSOCIATED WITH SONAR®

As a manufactured chemical that is released into the environment, Sonar® has been extensively evaluated for non-desired impacts in aquatic ecosystems. Much of this testing and evaluation has been reviewed as a facet of the NYS registration process, which resulted in the registration of Sonar® SRP in NYS, limiting its application to waters greater than two feet in depth. The registration process also resulted in the issuance of a Special Local Need (SLN) registration limiting the use of Sonar® A.S. to reduced application rates (50 ppb or less) for the control of Eurasian watermilfoil (Myriophyllum spicatum L.). However, as supported by extensive toxicological tests conducted during the product development and

FIFRA registration process, no adverse impacts have been identified which are expected to result from the presence of fluridone at or below the NYS unspecified organic compound concentration level of 50 ppb.

The EPA has designated an acceptable residue level for fluridone in potable water at 0.15 ppm (150 ppb) (USEPA, l986a). Independent studies have reported that fluridone has a very low level of toxicity to zooplankton, benthic macroinvertebrates, fish, and wildlife (Parka et al., 1978; McCowen et al., 1979; Arnold, 1979, and Grant et al., 1979). Arnold (1979) reported that fluridone is a safe, slow-acting herbicide that provides control of selected aquatic macrophytes, without impacting phytoplankton, zooplankton, benthic organisms or fish. Hamelink et al. (1986) concluded that fluridone is not expected to have adverse effects on the assortment of fish and invertebrates utilized in their study or on similar nontarget aquatic organisms. Furthermore, the potential for impacts can be reduced through the application considerations to maximize target selectivity as discussed in Section 4.5 and consideration of mitigation measures as discussed in Section 7.0. The following section discusses the potential impacts from the use of Sonar® in the water of NYS.

5.1 DIRECT AND INDIRECT IMPACTS TO NON-TARGET SPECIES

Sonar® is formulated as a selective aquatic herbicide for use in the management of unwanted aquatic macrophytes. As a chemical introduced into the environment, Sonar® has been evaluated during the registration process to determine potential adverse effects to non-target species. Direct impacts evaluated include toxicity, chronic changes in behavior or physiology, genetic defects or changes in breeding success or breeding rates for many test organisms. Indirect effects resulting from aquatic plant management may include changes in population size, changes in community structure or changes in ecosystem function. Both direct and indirect impacts can be evaluated at all stages of the life cycle of the non-target organism; though generally, the most sensitive stage of the organism (the young) is the period during which the organism is at greatest risk.

It should be noted that indirect impacts are often positive. For example, by controlling an exotic weed with Sonar®, the lake manager can facilitate the restoration of the native plant community. These desired changes in the community structure could be construed as an “impact”. The connotation of negative must be examined in light of the management objectives for the use of the product in the waterbody. Additionally, the balance of potential impacts must he considered in relation to the potential impacts from the presence of an exotic nuisance weed in an aquatic environment. The prevention of long-term impacts caused by unwanted aquatic plants may offset a potential short-term impact of the management program. Again, this issue should be evaluated for the waterbody of concern.

The direct toxicity of fluridone-based herbicides has been assessed using laboratory toxicity tests. The results of tests referenced in this section will be characterized according to the risk phases established by Christenson (1976) as follows:

EC or LC_{50}	Classiflcation
< 1 mg/1	Highly Toxic
1 - 10 mg/1	Moderately Toxic
10 - 100 mg/1	Slightly Toxic
100 - 1,000 mg/l	Practically Non-toxic
> 1,000 mg/1	Insignificant Hazard

Note: EC = Effective Concentration
LC_{50} = Concentration Considered to be Lethal to 50% of the Test Population

The following results should be considered in comparison to the 0.05 ppm concentration of fluridone allowed under the NYS drinking water concentration limit for all chemical compounds not specifically identified in the standards in waterbodies of NYS.

5.1.1 Macrophytes and Aquatic Plant Communities

Impacts to non-target rnacrophytes will be dependent on the sensitivity of that rnacrophyte to Sonar® at the application rate utilized (less than 50 ppb or 0.05 ppm), time of year of application, and use rate. Table 5-1 and Section 4.5.4 discuss those aquatic plants considered to be sensitive to Sonar® and fluridone. The loss of non-target plants within the aquatic plant community could alter the quality of functions that the vegetative community serves in the aquatic ecosystem. Loss of certain species from the community could alter the available habitat for fish species. The thinning of the macrophyte community could reduce the amount of refuge available to prey species and enhance the success of predators such as smallmouth bass. Such changes could benefit the fishery by altering the size distribution of the fishery (Andrews, 1989).

Lillie and Budd (1992) and Pullman (1993) suggest that in plant communities where Eurasian watermilfoil is in its pioneer stage of invasion or in heterogenous communities where watermilfoil is a component, habitat functions and values of this plant are considered to be comparable with native plant species. Therefore, the control of Eurasian watermilfoil in such communities could positively or negatively impact the associated fish community by temporarily reducing needed cover, shelter and food sources. However, it should be recognized that, once established, Eurasian watermilfoil is opportunistic and aggressive and demonstrates an ability to grow faster than and displace native plants (Pullman, 1993; Madsen, 1991b). The value of the fishery will then be degraded by loss of plant diversity resulting from excessive Eurasian watermilfoil growth.

TABLE 5-1

SENSITIVITY OF SUBMERGED AND FLOATING MACROPHYTE SPECIES TO SONAR APPLIED TO MICHIGAN LAKES

The sensitivity of common rnacrophyte species to Sonar when applied as whole lake treatments at rates used for the selective control of Eurasian watermilfoil and curlyleaf pondweed during the year of application and the year following application.

Common Name	Scientific Name	Response During Year of Application[1]	Response Following Year of Application[1]
Watershield	*Brasenia schreberi*	4	2
Fanwort	*Cabomba caroliniana*	5	?
Coontail	*Ceratophyllum demersum*	4-5	2
Charoid Algae	*Chara* spp. & *Nitella* spp.	1	2
Elodea	*Elodea chanadensis*	5	5
Water Stargrass	*Heteranthera dubia*	1	1
Northern Watermilfoil	*Myriophyllum sibiricum*	5	3
Eurasian watermilfoil	*Myriophyllum spicatum*	5	0
Watermilfoil	*Myriophyllum derticillatum*	3	3

Naiad	*Najas* spp.	4	2
Spatterdock	*Nuphar* spp.	4	2
Waterlilly	*Nymphaea* spp.	4	2
Broad Leaf Pondweed	*Potamogeton amplifolium*	3-4	2
Curlyleaf Pondweed	*Potamogeton crispus*	5	1-5
Illinois Pondweed	*Potamogeton illinoenis*	3-4	2
Sago Pondweed	*Potamogeton pectinatus*	4	1
Robin's Pondweed	*Potamogeton robbinsii*	1	3
Bladderwort	*Utricularia* spp.	1	3
Wild Celery	*Balliseria americana*	2-5	3

• The range of responses is related to the timing of the Sonar application.

TABLE 5-1 (CONTINUTED)

Response During Year of Application:

1 = Production or Total Distribution Increased
2 = Production or Total Distribution Slightly Increased
3 = No Impact on Plant Production or Distribution
4 = Production or Total Distribution Slightly Decreased
5 = Production or Total Distribution Drastically Decreased

Response Following Year of Application:

0 = Production Virtually Eradicated by Previous Year Application
1 = Production or total Distribution Increased
2 = Production or Total Distribution Slightly Increased
3 = No Impact on Plant Production or Distribution
(Production and Distribution Presumed to be Similar to Time of Pre-Milfoil Invasion)
4 = Production or Total Distribution Slightly Decreased
5 = Production or Total Distribution Dramically Decreased

Source: D. Pullman, Personal Communication, 1993

to grow faster than and displace native plants (Pullman, 1993; Madsen et al., 1991b). The value of the fishery will then be degraded by loss of plant diversity resulting from excessive Eurasian watermilfoil growth.

Sonar® controls all species listed on the label at the federal label application rate of 150 ppb. The label also lists species that may be partially controlled or are not controlled at these rates. Andrews (1989) notes that at low concentrations, Sonar® is highly selective to Eurasian watermilfoil and curly leaf pondweed. In a series of lake treatments in Michigan in 1992 at Sonar® application rates ranging from 8 to 29 ppb, Eurasian watermilfoil and curlyleaf pondweed were completely removed from the aquatic plant communities (Pullman, 1993). Non-target impacts included temporary herbicidal symptoms in water lilies (Nymphea and Nuphar spp.) and coontail (Ceratophyllum demersum). Pullman (1993) did report that elodea (Elodea canadensis) is susceptible to Sonar® and was usually removed from the plant communities in the treated lakes. He did observe that some native broadleaf pondweeds (Potamogedon spp.) appeared to be moderately to highly susceptible to Sonar® at application rates of 15 to 20 ppb, if the application occurred in the latter part of April and the early part of May. However, Pullman noted that native flora reestablished itself within a year of application. The production of Chara increased dramatically in nearly all lakes during the season of application. Water stargrass (Heteranthera dubia) and bladderwort (Utricularia spp.) also increased in area cover during the season of application.

In another lake treatment in Michigan, Pullman (1990) reported that at a Sonar® application rate of 0.014 ppm, Eurasian watermilfoil and curly leaf pondweed were removed from the water column in 4 to 6 weeks. In that treatment, water lilies exhibited some Sonar® induced chlorosis. Coontail was heavily impacted by the treatment, but persisted until the end of the growing season. Illinois pondweed (Potamogedon illinoensis) and water stargrass (Heteranthera dubia) were not affected by the Sonar® application and succeeded in expanding their distribution into areas previously colonized by the exotic aquatic macrophytes.

In a review of 21 lake treatments in Michigan in 1992, Kenaga (1992) noted that Sonar® effectively removed Eurasian watermilfoil and curlyleaf pondweed at concentrations as low as 8 ppb, where water exchange was minimal. The lakes ranged in size from two to 600 surface acres. In many of these lakes, non-target species had been limited by almost monoculture populations of nuisance exotic macrophytes. Kenaga (1992) went on to report that Sonar® was moderately effective at controlling southern naiad (Najas guadalupensis) and coontail (Ceratophyllum demersum) at 20 ppb, but relatively ineffective at controlling fanwort (Cabomba spp.).

In his 1992 preliminary draft report, Kenaga also noted that Sonar® effectively removed non-target species from the treated lakes at concentrations above 12 ppb. Re reported that after twelve to sixteen weeks from 20 to 100% of tho native plant community had been removed in the 21 lakes. However, he also noted that the study had not been of sufficient duration to evaluate the longer term control effectiveness of Sonar®, and even stated that pondweed regrowth was observed in two lakes at the end of the study. He also stated that several factors contributing to the low amounts of remaining cover could vary from lake to lake and could include:

a. A lack of accurate knowledge of the lakes depth resulted in treatment with a higher concentration of Sonar® then planned.

b. Succeeding yearly treatments.

c. Poor initial non-target plant communities. Monotypic stands of Eurasian watermilfoil or curlyleaf pondweed will result in very low populations of native plants. Kenaga noted that in 11 lakes in which the submersed native plant community was reduced in cover by 90 to 100% after 14 to 16 weeks, the initial native plant community was sparse to very sparse in terms of species diversity and density prior to treatment.

As previously discussed, Pullman (1993) stated that regrowth of the native plant community nearly always returned within a year of application. This is further supported in Pullman (1994).

Kenaga (1992) also reported that the primary emergent vegetation effected by Sonar® were water lilies and cattails. Impacts to these species were primarily chlorosis and damage to plant foliage. However, even with damage or lost leaves, most water lilies were still observed to flower, indicating the continuing viability of the plant. Kenaga did note that emergent vegetation in lakes treated early in the season or in the 8 to 10 ppb range, experienced the least damage.

In an experimental lake treatment in Florida using both Sonar® A.S. and Sonar® SRP, hydrilla (Hydrilla verticillata) and Illinois pondweed (Potamogeton illinoensis) were the only two submerged aquatic macrophytes significantly impacted by the application. Coontail (Ceratophyllum demersum), southern naiad (Najas quadalupensis), bladderwort (Ultricularia spp.) and eelgrass (Vallisneria americana) were unaffected by the Sonar® application

Fluridone has the potential to impact terrestrial plants through the use of water containing fluridone for irrigation purposes. Recommended time frames for delaying use of treated water for irrigation are summarized on the Sonar labels.

5.1.2 Algal and Planktonic Species

Sonar® is not considered to be effective as an algicide (product label). Pullman (1993) reported that chara rapidly spreads in the littoral zone of Michigan lakes following Sonar® use for removal of Eurasian watermilfoil or curlyleaf pondweed. Filamentous algae and Nitella increased in Lake Sompson, Florida, following treatment with Sonar® (Hinkle 1985). Parka et al. (1978) noted that fluridone did not appear to adversely affect desirable phytoplankton at treatment concentrations of 0.3 and 0.1 ppm. They did report some temporary reductions in less desirable blue-green phytoplankton species such as Anabaena and Anacystis. Similarly, Kammarianos et al. (1989) reported the elimination of bloom causing blue-green algae (Cyanophyceae) following the treatment of a Greek pond with Sonar® A.S., which resulted in a water concentration of 0.042 ppm of fluridone. However, diatoms and other phytoplankton species (Diatomaceae, Chlorophyceae, Dinophyceae and Englenineae) increased after Sonar® use. The authors concluded that no detrimental effects were apparent. Struve et al. (1991) reported no sufficient reduction in phytoplankton densities when two ponds in Alabama were consistently exposed to a fluridone concentration of 0.125 ppm. Fluridone as an aqueous solution, when applied at the exaggerated rate of 1.0 ppm resulted in the reduction of zooplankton species, while an application rate of 0.3 ppm did not produce any effects in the zooplankton community (Arnold, 1979). In the 1.0 ppm treated pond zooplankton populations returned to pretreatment levels within 43 days. Arnold reported similar trends in the phytoplankton population.

Kenaga (1992) reported that Chara expanded almost exponentially following the removal of submersed macrophytes in most lakes that he surveyed in Michigan. He also noted a perceived improvement in water clarity. While not scientifically documented, Kenaga reported that the possible reason for the improvement in water clarity was the increased growth in Chara.

5.1.3 Fish, Shellfish and Aquatic Macroinvertebrates

USEPA (1986a) summarizes the data developed from exposure of aquatic organisms in standard static water LC_{50} toxicity tests. Following exposure of Daphnia magna for 48 hours, the concentration of fluridone calculated to product an acute response in 50% of the test population was 6.3 ppm. Following exposure of rainbow trout (Salmo gairdneri) and bluegill (Lepomis macrochrius) for 96 hours, the concentration of fluridone calculated to produce a lethal response in 50% of the test population was 11.7 ppm and 12 ppm, respectively.

USEPA (1986a) also lists a Maximum Acceptable Toxicant Concentration (MATC) of greater than 0.48 ppm, but less than 0.96 ppm, for exposure of fathead minnow fry (Pimephales promelas) to fluridone, indicating that no treatment related effects on fathead minnow reproductive measures were observed at or below 0.48 ppm. Struve et al. (1991) observed that fish abundance and community structure remained unchanged in ponds exposed to a fluridorie concentration level of 0.125 ppm.

Parka et al. (1978) reported that at the exaggerated rate of 1.0 ppm of fluridone in water, the total numbers of benthic organisms were significantly reduced when compared to a control population. They also noted that 0.3 ppm of fluridone in water did not significantly reduce total numbers of benthic organisms. Fluridone as an aqueous solution, when applied at the rate of 1.0 ppm resulted in the reduction of populations of the amphipod Hyalella azeteca while an application rate of 0.3 ppm did not result in the reduction of amphipod populations (Arnold, 1979). Naqvi and Hawkins (1989) reported Sonar LC_{50} values of 12.0 ppm, 8.0 ppm, 13.0 ppm and 13.0 ppm for the microcrustaceans Diaptomus sp., Eucyclops sp., Alonella sp., and Cypria sp., respectively.

Hamelink et al. (1986) conducted extensive acute and chronic toxicity tests on numerous fish and invertebrate organisms. For invertebrates, they noted an average 48-hour or 96-hour LC_{50} or EC_{50} (depending on the organisms) fluridone concentration of 4.3 ± 3.7 ppm. The representative invertebrates used in the study included amphipods (Gammarus pseudolimnaeus), midges (Chironomus pulmosus), daphnids (Daphnia magna), crayfish (Orconectes immunis), blue crabs (Callinectes sapidus), eastern oysters (Crassostrea virginica), and pink shrimp (Penaeus duroarum). For fish, they noted an average 96-hour LC_{50} fluridone concentration of 10.4 ± 3.9 ppm. The representative fish used in their study included rainbow trout (Salmo gairdneri), fathead minnows (Pimephales promelas) channel catfish (Ictalurus punctatus), bluegills (Lepomis macrochirus) and sheepshead minnows (Cyprinodon variegatus).

In the chronic toxicity tests conducted by Hamelink et al. (1986), no effects were observed in daphnids, amphipods, and midge larvae at fluridone concentrations of 0.2, 0.6, and 0.6 ppm, respectively. They reported that channel cattish fry exposed to fluridone concentrations of 0.5 ppm were not significantly affected. Catfish fry growth was reported as reduced at fluridone concentrations of 1.0 ppm. They also reported that chronic exposure of fathead minnows to mean concentrations of 0.48 ppm did not produce adverse effects. Results from Hamelink et al. (1986) indicated that fluridone concentrations of 0.95 and 1.9 ppm resulted in reduced survival of fathead minnow within 30 days after hatching.

5.1.4 Avian Species

USEPA (1986a) notes that acute toxic effects were not observed in bobwhite quail (Colinus virginianus) following the oral administration of a dose concentration of 2000 mg/kg of fluridone. USEPA considers this to be a slightly toxic response. Avian 8-day dietary studies for the bobwhile quail and the mallard ducks (Anas platyrhynchos) resulted in no mortality at 5000 ppm fluridone in the bird's food ration. (USEPA, 1986). USEPA further reported that no reproductive impairments in bobwhite quail or mallard ducks were observed following dietary. exposure of up to 1000 ppm.

5.1.5 Mammals

Metabolism and distribution tests have shown that fluridone is, absorbed and excreted in the feces within 72 hours of oral administration within rats. Acute toxicity studies have shown that the LD_{50} for a rat (Rattus norvegicus) exposed through the oral pathway to technical grade fluridone is greater than 10,000 ppm. Ingestion of Sonar® A.S. by rats resulted in no mortality when administered at 0.5 ml/kg. The LD_{50} for a mouse (Mus musculus) exposed through the oral pathway to technical grade fluridone is greater than 10,000 ppm. The LD_{50} for a cat (felis domesticus) exposed through the oral pathway to technical grade fluridone is greater than 250 ppm. The LD_{50} for a dog (Canis familiaris) exposed through the oral pathway to technical grade fluñdone is greater than 500 ppm (Elanco, 1981).

In 90-day subchronic feeding studies, no treatment-related effects were noted in rats at dietary doses of 330 ppm fluridone or in mice at dietary doses of 62 ppm fluridone. No toxic effects were observed in dogs at dietary doses of fluridone of 200 mg/kg/day. In one-year feeding studies, a dietary level of fluridone o 200 ppm did not produce toxic effects in rats and a 100 ppm dietary level did not produce toxic effects in mice. The administration of 150 mg/kg/day of fluridone to dogs for one year did not produce toxicological effects. Two-year feeding studies resulted in no evidence of carcinogenicity. In reproductive studies, fluridone was not teratogenic to rats at 200 mg/kg/day or rabbits at 750 mg/kg/day when administered during the organogenesis phase of gestation. Three successive generations of rats maintained on diets containing 2000 ppm of fluridone showed no impairment of fertility, liveborn litter size, gestation length or survival, progeny survival, or sex distribution (Elanco, 1981). Table 5-2 summarizes the NOEL's identified in toxicological tests conducted on fluridone. NOEL (No Observed Effect Level) is the highest dose tested which did not produce effect in the test group. For relative comparison of toxicity values, a listing of the toxicity of some common chemicals follows in Table 5-3.

5.1.6 Reptiles and Amphibians

Toxicity tests have not been conducted on any reptile or amphibian species, nor have they been required under the FIFRA process. Qualitative observations made by Arnold (1979) in field tests of fluridone in an aqueous solution at application rates of up to 1.0 ppm noted that frogs (Rana spp.), watersnakes (Nerodia spp.), and softshell turtles (Trionyx spp.), were not obviously impacted by the herbicidal application.

5.1.7 Federal and State Listed Rare, Threatened, and Endangered Species

Endangered species are those organisms faced with extinction in all or much of their distribution, Threatened species are those organisms chat seem likely to become endangered. Rare species are those organisms which have widely scattered populations or are few in number. These organisms are rare for a variety of reasons, including changes in habitat (both natural and man made), at the extent of its geographical range and predation pressure. Federal identified species are listed under the 50 CFR § 17.11 and § 17.12. State listed species are identified in NYCRR § 193.3.

Acute aquatic toxicity values and MATC's suggest that potential hazards to aquatic organisms would only be seen at concentrations higher than labeled application rates. This is particularly true in New York, where the maximum label rate for use of Sonar® A.S. is 0.05 ppm in treated. water. It should also be noted that Sonar® labeling states that "to avoid impact on threatened or endangered aquatic plant or animal species, users must consult their State & Game Agency or the U.S. Fish and Wildlife Service before making applications". Identification of any rare, threatened or endangered species should be made as part of a permit application. A complete listing of threatened and endangered plant species in NYS is presented in Appendix B.

5.1.8 Biodiversity Sites

Information on the known location of rare species and significant natural communities can by obtained from the NYS Natural Heritage Program, which maintains a database on those resources. A determination of whether the proposed location of a Sonar® application would occur in one of these areas may be made through the Natural HeritageProgram as part of the evaluation of a permit application.

TABLE 5-2

SUMMARY OF NOEL'S IDENTIFIED IN TOXICOLOGICAL RESEARCH CONDUCTED ON FLURIDONE

FLURIDONE STUDIES	NOEL RESULTS
90-day feeding study	53 mg/kg/day in the diet
90-day mouse feeding study	9.3 mg/kg/day in the diet
90-day dog feedng study	200 mg/kg/day administered orally
1-year rat feeding study	9.4 mg/kg/day in the diet
1-year mouse feeding study	11.4 mg/kg/day in me diet
1-year dog feeding study	150 mg/kg/day
2-year rat chronic feeding/oncogenicity studies	8.5 mg/kg/day in the diet No evidence of carcinogenicity at any feeding level
2-year mouse chronic feeding/oncogenicity studies	11.6 mg/kg/day in the diet No evidence of carcinogenicity at any feeding level
Modified Ames test	Negative at level of compound solubility
Unscheduled DNA repair synthesis assay	Negative in cultured rat hepatocytes at 1 micromole/ml
Sister chromatid exchange assay	Negative at an intraperitoneal dose of 500 mg/kg in Chinese hamster bone marrow
Dominant lethal test in male rats	Negative at an oral dose of 2,000 mg/kg
Rat teratology study	200 mg/kg/day
Rabbit teratology study	750 mg/kg/day
3-generation rat reproduction study	121 mg/kg/day in the diet

Notes: NOEL = No Observed Effect Level
mg/kg/day = milligram/kilogram/day
mg/kg = milligram/kilogram
micromole/ml = micromole/rnilliliter

Source: NYSDOH, 1986

TABLE 5-3

APPROXIMATE TOXICITY VALUES FOR OTHER COMMON CHEMICALS RELATIVE TO SONAR

COMPOUND	LD50
Technical Grade Fluridone	>10,000 mg/kg*
Table Salt	3,000 mg/kg
Vitamin A	2,000 mg/kg
Asprin	1,000 mg/kg
Caffeine	164 mg/kg
Nicotine	53 mg/kg

• For exposure to rats via the oral pathway

5.2 POTENTIAL FOR IMPACT FROM THE ACCUMULATION/DEGRADATION OF TREATED PLANT BIOMASS ON WATER QUALITY

The rapid defoliation of aquatic plants in the water column can negatively impact Dissolved Oxygen (DO) levels in the waterbody as a result of the biological degradation of the organic material. This can impact the fish populations in the surrounding area. It is not expected that this event would occur foflowing the use of Sonar®. Sonar® is a slow acting systemic herbicide which can take 30 to 60 days to produce its herbicidal effects in the target population. This results in a slow addition of organic material into the water column. Various researchers (Parka et al., 1978 and Struve et al., 1991) reported that Sonar® applications of up to 0.125 ppm have not resulted in significant decreases in DO content. In field tests conducted by Arnold (1979), fluridone in an aqueous solution at application rates of up to 1.0 ppm did not change water quality parameters as measured by DO, pH, Biological Oxygen Demand (BOD), color, dissolved solids, hardness, nitrate, specific conductance, total phosphates, and turbidity. Osborne et al. (1989) and West et al. (1990) also did not identify any changes in DO levels following application of Sonar®

As discussed in Section 4.8.1, several authors (West et al., 1979 and Langeland and Warner, 1986) reported that low concentrations of fluridone are released back into the water system as the plant material degrades. Langeland and Warner (1986) noted an increase from 20.9 ppb to 28.9 ppb at day 51 of their degradation trial at a pond in Virginia. However, this increase is not to a level considered to be detrimental to fish population and is taken into account with regards to the overall degradation profile of fluridone which is discussed in Section 4.0. As such, the rerelease of fluridone into the water column from decaying plant material is not considered to be a potential for ecological concern.

5.3 IMPACT OF RESIDENCE TIME OF SONAR® IN THE WATER COLUMN

As discussed in the previous sections, Sonar® is a slow acting systemic herbicide that degrades with an average half-life of approximately 20 days in the water column. The chemical is designed to remain in the water column long enough to produce its effects and the application concentrations of fluridone are below those considered to be toxic to most aquatic organisms. Therefore, it is not anticipated that the residence time in the water column would alter the projected impacts that have been discussed.

5.4 RECOLONIZATION OF NON-TARGET PLANTS AFTER CONTROL OF TARGET PLANTS IS ACHIEVED

It is expected that following the reduction of coverage of nuisance macroPHYTES such as Eurasian watermilfoil and curlyleaf pondweed which are sensitive to low-level application rates of Sonar®, that the more tolerant native aquatic macrophyte species would expand into the vacated niches. Pullman (1993) supports that assumption based on observations of Sonar® application in lakes in Michigan. Certain species such as water stargrass, Chara, Nitella, bladderwort, and Illinois pondweed may actually expand enough to become a nuisance the year after Sonar® application. Kenaga (1992) reported exponential growth in Chara in most of the 21 lakes he surveyed in Michigan that were treated with Sonar®. Dechoretz (1991) reported that regrowth by pondweeds, coontail and other native plants occurred generally within six to eight months following treatment of ponds in California with Sonar® A.S. and Sonar® SRP at the labeled application rates (0.15 ppm).

5.5 IMPACTS ON COASTSL RESOURCES

As noted in Section 5.1.3, the use of Sonar® herbicides at the recommended application rates is not likely to result in any adverse toxicological effects to marine species. The likelihood of any effects is also reduced by the probability of heavy dilution of any herbicide reaching the water column due to wave, current, and tidal activity.

If the use of Sonar® herbicides is proposed to be located within the NYS Coastal Zone and is determined to require federal licensing, permitting, or approval, or involves federal funding, then the action would be subject to the NYS Coastal Zone Management Program (19 NYCRR Section 600). This determination, would be required during the preparation of an individual permit application. It should be noted that the label for Sonar® SRP states that it should not be applied in tidewater/brackish water and the SLN label for Sonar® A.S. allows its use only in freshwater ponds, lakes, and reservoirs.

6.0 POTENTIAL PUBLIC HEALTH IMPACTS OF SONAR

6.1 BRIEF OVERVIEW OF FLURIDONE TOXICITY

USEPA (1986a) has reported that technical grade fluridone, as used in manufacturing, is in Category IV for acute oral effects in the rat and is moderately toxic through acute inhalation exposure. Eye irritation for technical fluridone potential has been demonstrated as moderate to severe (Category III and Category II). Both the aqueous suspension and pellet formulations are in Category III for oral, dermal, skin, and eye irritation effects. Consequently, Sonar® A.S. and Sonar® SRP labels bear a “Caution” signal word.

Metabolism and distribution tests have shown that fluridone is absorbed and excreted in the feces within 72 hours of oral administration to rats. Acute toxicity studies have shown that the LD_{50} for a rat (Rattus norvegicus) exposed through the oral pathway to technical grade fluridone is greater than 10,000 mg/kg. Administration of Sonar® 4 A.S. to rats at 0.5 ml/kg did not provoke a lethal response. The LD_{50} for mice (Mus musculus) exposed through the oral pathway to technical grade fluridone was greater than 10,000 mg/kg. The LD_{50} for cats (Felis domesticus) exposed through the oral pathway to technical grade fluridone was greater than 250 mg/kg. The LD_{50} for dogs (Canis familiaris) exposed through the oral pathway to technical grade fluridone was greater than 500 mg/kg (Elanco, 1981).

In 90-day subchronic feeding studies, no treatment-related effects were noted in rats at dietary doses of 330 mg/kg or in mice at dietary doses of fluridone of 62 mg/kg. No toxic effects were observed in dogs at dietary doses of fluridone of 200 mg/kg/day. In chronic toxicity studies, dietary levels of fluridone of 200 mg/kg did not produce toxicological or carcinogenic effects for either a one or two year test period. In reproductive studies, fluridone was not teratogenic to rats at 200 mg/kg/day or rabbits at 750 mg/kg/day when administered during the organogenesis phase of gestation. Three successive generations of rats maintained on diets containing 2000 mg/kg of fluridone showed no impairment of fertility, liveborn litter size, gestation length or survival, progeny survival, or sex distribution (Elanco, 1981).

6.2 NYS DRINKING WATER STANDARD

The drinking water standard established in New York State for any organic chemical contaminant not specifically identified in the standards is either 5 ppb or 50 ppb, depending on the chemical structure. Based on its chemical structure, the drinking water standard for fluridone is 50 ppb. Pursuant to the SLN, application of Sonar® A.S. is limited to application rates of 50 ppb. The release of fluridone from the pellet formulation (Sonar® SRP) will not result in fluridone concentrations exceeding 50 ppb at the labeled application rate. No adverse health effects have been identified at fluridone concentrations of 50 ppb or less. Kim (1992) states that at the 50 ppb application rate, no restrictions are necessary on the use of Sonar® A.S. in water bodies that serve as sources of potable water, beyond not allowing swimming for 24 hours and those restrictions on the federal label. Kim does recommend for Sonar® SRP that application should be prohibited in waters less than 2 feet deep. USEPA (1986a) has designated an acceptable residue level for fluridone in potable water at 0.15 ppm (150 ppb). Sonar® cannot be applied within one-fourth mile (1320 feet) from any functioning potable water intake.

7.0 MITIGATION MEASURES TO MINIMIZE ENVIRONMENTAL AND HEALTH IMPACTS FROM SONAR

Mitigation measures describe guidelines to mitigate or lessen the potential for impacts from the use of Sonar® in the waters of NYS. While no impacts to humans are expected from the use of Sonar® in the waters of NYS, there is the potential for some ecological effects. The mitigation measures described in this section will reduce, or mitigate that potential for ecological effects, without reducing the efficacy of the product.

7.1 USE CONTROLS

When the aquatic plant management objective is to control Eurasian watermilfoi1, while minimizing impacts to other aquatic macrophytes, Sonar® may be used early in the season. As was discussed in Section 3.5.1, Eurasian watermilfoil is essentially evergreen and begins to grow rapidly at the beginning of the growing season. This enables this plant to develop significant biomass before native macrophyte species begin growing (Smith and Barko, 1990). The use of Sonar® early in the growing season would target Eurasian watermilfoil, while minimizing the impact on other aquatic vegetation.

For removal of Eurasian watermilfoil with minimal impact on other species, it is suggested that Sonar® products be uniformly applied across the entire area to be treated. Applicators should follow an application pattern that minimizes concentration of the product in local areas. When making lake-wide treatments it is recommended that application rates, calculated as ppb of fluridone be based only on the water volume in which mixing is expected to occur. Calculations should be based on water volume in the epilimnion above any deep water areas below the metalimnion or thermocline.

7.2 LABEL INSTRUCTIONS

The USEPA approved label for Sonar® SRP and the NYSDEC Special Local Need supplemental label for Sonar® A.S. list several general use precautions for the two products. The sale of Sonar® A.S. solely under the USEPA approved label is not permitted in NYS. The use is only allowed in conjunction with the SLN label. The SLN label for Sonar® A.S. specifies the use of this product for Eurasian watermilfoil only. Label use precautions and directions include the following:

1) Before applying the product, notification of and approval of the NYS Department of Environmental Conservation is required, either by an aquatic permit issued pursuant to ECL Section 15.0313(4) or issue of purchase permits for such use

2) In lakes and reservoirs, do not apply Sonar® A.S. within one-fourth mile (1320 feet) of any functioning potable water intake. Existing potable water intakes which have been disconnected and are no longer in use, such as those replaced by connections to potable water wells or a municipal water system, are not considered to be functioning potable water intakes.

3) Irrigation with Sonar® treated water may result in injury to the irrigated vegetation.

4) Follow use directions carefully so as to minimize adverse effects on nontarget organisms. In order to avoid impact on threatened or endangered aquatic plant or animal species, users must consult their State Fish arid Game Agency or the U.S. Fish and Wildlife Service before making applications.

5) Do not apply in tidewater/brackish water.

6) Lowest rates should be used in shallow areas where the water depth is considerably less than the average depth of the entire treatment site, for example, shallow shoreline areas.

7.3 RELATIONSIP TO THE NYS DRINKING WATER STANDARD

The drinking water standard established in New York State for all chemical compounds not specifically identified in the standards is 50 ppb. No adverse health effects have been identified at fluridone concentrations of 50 ppb or less. Kim (1992) states that at the 50 ppb application rate, no restrictions are necessary on the use of Sonar® AS in water bodies that serve as sources of potable water. As discussed in Section 4.4, Sonar® is effective as a selective systemic herbicide at the application rate of 50 ppb or less.

7.4 RULEMAKING DECISIONS

As of April 7, 1993, all pesticides labeled for use in aquatic settings were classified as restricted use products by regulation of the New York State Department of Environmental Conservation. Under this regulation, 6 NYCRR Parts 325 and 326. The use of aquatic pesticides, including Sonar® A.S., and Sonar® SRP, is limited to persons privately certified, commercially certified in Category 5, or possessing a purchase permit for the specific application that is proposed. Additionally, only those persons who are certified applicators, commercial permit holders, or have a purchase permit may purchase aquatic use pesticides.

With respect to fluridone, the regulations place the following restrictions on its use:

1. Aqueous suspension formulations may be applied at application rates not to exceed 50 ppb.

2. Pellet formulations may be applied to water two feet or greater in depth.

3. Swimming is not allowed in treated waters for 24 hours following application.

The effect of these rules will be to reduce the potential for risks to public health and the environment.

Under Part 327, a site specific permit will be required for the use of Sonar® in the waters of NYS, unless the waterbody is a privately-owned, no-outlet pond. The permit is issued through the NYSDEC. Potential permit applicants are cautioned to utilize the most recent product label for the development of their permit application. The applicants for the permit are required to be a riparian owner, or a lessee of a ripairan owner, or association of such persons. The applicant is required to submit the permit on a form provided by the NYSDEC. The information required for the application includes;

1. A scale drawing or map, including depth soundings adequate to determine: the size and depth of the treatment area; the concentration of the chemical within the area and the conformity to the limitations set forth in the regulations; the location and type of submerged and emergent weed beds; the location of water users relative to the area and along the outlet; and any further information required by the permit-issuing official.

2. Applications that involve public water supply waters or their tributaries will be referred to the State DOH for approval before the permit is issued.

3. The applicant must certify: that the listed chemical will be employed in conformance with all conditions specified in the permit issued; that the applicant obtained agreements to the treatment from water users whose use may be restricted as set forth in the application; that the applicant agrees that the issuance of the permit is be based on the assumed accuracy of all statements presented by him; that the applicant is legally responsible for damages resulting from the application of the chemical, or from the inaccuracy of any computations or from improper application of the chemical; and that the applicant assumes full legal responsibility for the accuracy of all representations made in obtaining approvals or releases, and for any failure to obtain approval or releases from the persons likely to be adversely affected.

A full copy of the Part 327 regulation is contained in Appendix C to this GEIS.

The use of SONAR within any jurisdictional wetland in the Adirondack Park is a regulated activity requiring a wetland permit from the APA pursuant to 9 NYCRR Part 578. The Agency's permit application requests information similar to that required by the NYSDEC, however additional details on the identification of all plant species including rare or endangered and their relative density within the treatment area will be necessary.

7.5 SPILL CONTROL

Care should be taken to use Sonar® properly and in accordance with the approved labels. Any leaks or spills should be promptly addressed. Liquid spills on an impervious surface should be cleaned up using absorbent materials and disposed of as waste. Liquid spills on soil may be handled by removal of the affected soil, and disposal at an approved waste disposal facility. Leaking containnrs should be separated from non-leaking containers and either the container or its contents emptied into another container. Spills of granular material should be promptly picked up, placed in a container and used according to label directions or disposed of in a proper manner at an approved waste disposal facility.

7.6 OTHER MITIGATION CONSIDERATIONS

In addition to the above mentioned activities, the following measures may be considered to further reduce, or mitigate any potential for environmental effects, without reducing the efficacy of the product.

7.6.1 Timing of Application

The potential for non-target impacts may be mitigated by the selection of an optimum time for application. It is recommended that Sonar® be applied as early in the growing season as possible. Eurasian watermilfoil initiates productivity and metabolic activity at an earlier time than native plants (Smith and Barko, 1990). As a result of those growth characteristics, an early season application is recommended. This would allow for treatment of Eurasian watermilfoil while the remaining plant community is still dormant. Based on observations made in Michigan, Pullman (1993) noted that several broadleaf pondweeds may be moderately to highly susceptible to fluridone at application rates of 15 to 20 ppb. if the application occurs as these plants begin to grow.

Additionally, early season application would be conducted while the water is relatively cold. Dissolved Oxygen levels during that time of the year are generally high, thereby mitigating any possibility of impacts to fisheries. Also, recreational use of water during that time frame would be limited (Pullman, 1994).

7.6.2 Application Techniques

The choice of Sonar® SRP or Sonar® A.S. could serve as a means of mitigating the potential for impacts to non-target macrophytes. The selection of Sonar® SRP versus Sonar® A.S. should be based on the management objectives of the aquatic macrophyte control program for the particular waterbody. The selection of one formulation or the other is related to maintaining an appropriate concentration of fluridone for a sufficient amount of time to allow for uptake by the target macrophyte. Generally, Sonar® SRP is more appropriate for moving water because it releases fluridone over a longer period of time than the A.S. formulation. This will allow for a longer exposure time than the liquid formulation which would tend to be more rapidly diluted by untreated water.

Sonar SRP is recommended when applied while the target submerged plants are low growing in the water column and where bottom sediments are sands or other firm substrates

SONAR® A.S. is recommended where target submerged plants have grown to near the water surface. Sonar A.S. performs well when applied over soft muck or organic sediments.

8.0 UNAVOIDABLE ENVIRONMENTAL IMPACTS IF USE OE SONAR IS IMPLEMENTED

As detailed in Section 6.0, the use of Sonar® has been evaluated during federal and New York State registration process and in this GEIS for various impacts to non-target organisms in the aquatic setting. There are several unavoidable impacts that will occur when Sonar® is used in the waters of NYS to manage unwanted aquatic macrophytes such as Eurasian watermilfoil. It is important to note that the mitigation approaches described in Section 7.0 will lessen the magnitude and extent of those impacts. Those impacts are:

1. Impact to Habitat

   When Sonar® is introduced into a waterbody, it will result in the death of the target macrophytes. Once these target macrophytes have dropped out of the water column, there will be a period of time before the native non-target macrophytes reestablish themselves in the vacant niches. While the non-target species will reestablish themselves as detailed in Section 5.4, the process is not immediate. During that period of time, the aquatic macrophyte community will be reduced in size.

2. Impacts to Non-target Species

   A review of the literature indicates that there are native macrophytes which would be impacted to some extent by the use of fluridone in a waterbody. This has been detailed in Section 5.1.1. However, the literature indicates that a plant community composed of native plant species will become reestablished during the season following Sonar® use.

3. Possible Reinfestation

   In areas of significant water flow, such as lake inlets, Eurasian watermilfoil and other target plants may not be sufficiently controlled due to the dilution of applied Sonar® with untreated water. The reinfestation of Eurasian watermilfoil may occur via the dispersal means described in Section 3.3.1. This may necessitate the utilization of alternative means of controlling Eurasian watermilfoil in those areas of rapid water movement.

9.0 ALTERNATIVES TO SONAR®

This section details the various alternatives to the proposed action. The other alternatives include the no-action alternative to the use of Sonar® (which entails the lack of any aquatic macrophyte control measure, except as specified), chemical alternatives to Sonar®, mechanical alternatives to Sonar®, biological alternatives to Sonar®, and various other options. The no-action alternative does not preclude the ability of an applicant to apply for a permit for the use of those products described in the Final Programmatic Environmental Impact Statement on Aquatic Vegetation Control (NYSDEC, 1981a). Each of the possible alternatives will be evaluated from the standpoint of efficacy, positive and negative environmental impacts, and relative costs. The choice of a particular alternative over the proposed use of Sonar® should be based on the management objectives for the waterbody and the specific characteristics of the problem.

9.1 NO-ACTION ALTERNATIVE

In the no-action alternative, aquatic macrophyte control measures which could be utilized in the waterbodies of potential concern would be those chemical and mechanical means identified in the Final Programmatic Environmental Impact Statement on Aquatic Vegetation Control (NYSDEC, 1981a). Under the no-action alternative, the use of Sonar® is not considered for the control of the growth and spread of the target macrophytes in the waterbodies of concern. In this scenario, the only controlling measures, other than natural fluctuations in the plant populations, would be those activities presently permitted in NYS waterbodies. Without any controlling measures, the spread of invasive weeds such as Eurasian watermilfoil could result in significant modifications of the native aquatic habitat of a particular waterbody. Uncontrolled invasive macrophytes produce seeds and/or other reproductive parts that can be spread to other aquatic sites.

As discussed in Section 3.3.1, a large number of researchers have documented the negative impact of the introduction of Eurasian watermilfoil in a waterbody (Aiken et al., 1979; Lonsdale and Watkinson, 1983; Keast, 1984; Nichols and Shaw, 1986; and Smith and Barko, 1990). Madsen et al. (1991a) documented the decline of native macrophytes in a New York lake as a result of the invasion of Eurasian watermilfoil. Without any controlling measures, Eurasian watermilfoil can potentially modify the native plant community in a significant manner. Eurasian watermilfoil, once it has begun to form its characteristic canopy, will displace non-canopy forming native species. The result of the typical growth pattern of Eurasian watermilfoil is to form dense monotypic stands.

Pullman (1993) concluded that Eurasian watermilfoil is supportive of fish populations during its initial expansion stages in a waterbody. However, he goes on to note that once Eurasian watermilfoil begins to dominate the plant community and form its characteristic dense mats, the lack of plant species diversity and associated water quality impacts will reduce the quality of the habitat for fish. Nichols and Shaw (1986) reported that Eurasian watermilfoil provides beneficial cover for fish, unless the cover is so dense that stunting of fish growth from overcrowding results. Eurasian watermilfoil has been shown to provide a better habitat for fish (Kilgore et al, 1989) and invertebrates (Pardue and Webb, 1985) than open water. However, Dvorek and Best (1982) found that Eurasian watermilfoil had the poorest invertebrate fauna populations out of 8 aquatic macrophyte species that were examined. Keast (1984) noted that fish abundance was 3 to 4 times greater in mixed native plant communities than in a plant community dominated by Eurasian watermilfoil. Nichols and Shaw (1986) noted that Eurasian watermilfoil is poor food for muskrats and moose and fair food for ducks, which will eat its fruit.

Eurasian watermilfoil also impacts the recreational use a waterbody by interfering with swimming and boating, by reducing the quality of sport fisheries, and by reducing the aesthetic appeal of waterbodies (Newroth, 1985). Because of its mat forming characteristics, excessive growth of Eurasian watermilfoil (a primary target species for Sonar® may present a safety hazard to the recreational use of a waterbody. The mats may cover rocks, logs, and other obstructions that could damage moving boats or injure water skiers. Additionally, the mats may entangle swimmers, potentially resulting in drownings. Drownings as a result of entanglement in Eurasian watermilfoil mats have been documented in New York (Long et al., 1987). NYSDEC (1981) notes that the lack of vegetation control may result in economic loss to the state and may reduce water quality, hinder desired human usages, and present health hazards.

Keast (1984) noted that fish populations and their invertebrate prey species are reduced in dense mats of Eurasian watermilfoil. Excessive Eurasian watermilfoil growth will result in clogged industrial, potable and power generation intakes, lowered dissolved oxygen concentrations, and increased populations of permanent pool mosquitoes (Bates et al., 1985). Additionally, the failure to control an invasive species such as Eurasian watermilfoil can jeopardize uninfested lakes by increasing the likelihood of the spread of the plant (VDEC, 1993).

Under the no-action alternative, there is the potential for subsequent declines in Eurasian watermilfoil following the invasion of a particular waterbody by the plant. Smith and Barko (1990) note that the population growth patterns of Eurasian watermilfoil in many waterbodies often vary to a great extent over time and from location to location. A variety of hypotheses have been presented to explain these population declines. They include nutrient depletion, shading by phytoplankton, attack by parasites, climatic fluctuations, and long-term effects of aquatic weed control. (Carpenter, 1980). Smith and Barko (1990) note that declines have been documented in Wisconsin, British Columbia, and the Chesapeake Bay area. Painter and McCabe (1988) reported the decline and disappearance of Eurasian watermilfoil from several lakes in Ontario, Canada. No reason was confirmed for the disappearance, though circumstantial evidence indicated insect herbivory as the cause.

Carpenter (1980) reports that the period of peak abundance in these locations has ranged from approximately 5 to 10 years, with 10 years seen as the typical time frame. However, fluctuations in Eurasian watermilfoil populations are not generally predictive. In some areas, population fluctuations have been, limited to seasonal changes or have not been observed (Grace and Wetzel, 1978; Madsen et al., 1988a; Kimbel, 1982; Nichols and Shaw, 1986; and Madsen et al., 199lb). Pullman (1992) noted declines in several Michigan lakes; though the declines were generally short-lived and populations soon returned to pre-decline levels. FOLA (1994) noted that the decline of Eurasian watermilfoil populations in Cayuga Lake appeared to be associated with the spread of the European aquatic moth larva (Acentria nivea). As detailed in Section 3.3.1, the number of lakes throughout the northeastern United States in which Eurasian watermilfoil infestation has been observed is increasing.

Some research has shown that the failure to manage Eurasian watermilfoil in a waterbody can have financial impacts to the recreational use of the waterbody. In a socio-economic research study in an area of 8 lakes infested with Eurasian watermilfoil, BCMELP (1991) estimated a loss in several economic areas, including transportation, the restaurant industry, the accomodation sector, and the shopping sector. They projected that a no-action alternative to managing for Eurasian watermilfoil would result in a loss in revenues in 1990 of $85 million in the Okanagan Valley region of British Columbia, Canada (or 26.5% of 1989 revenues). They also predicted a loss of 1700 employment positions in the tourist industry and a loss in real estate values of $360 million in the region. However, these figures have not been verified by the British Columbia Ministry of Environment, Lands and Parks,

9.2 CHEMICAL ALTERNATIVES

NYSDEC (1981) presented an evaluation of various chemical alternatives to Sonar®. Generally, chemical herbicides are divided into two broad categories. Those categories include contact herbicides and systemic herbicides. Contact herbicides remove that part of the plant that they come in contact with. Plant regrowth typically occurs within a few weeks or months. Systemic herbicides are absorbed by the plant and translocated to the lower stem and root system, which results in longer term plant control. Because of the systemic nature of Sonar®, another submersible systemic herbicide would be its most logical chemical alternative.

NYSDEC (1990) notes that aquatic herbicides are chemicals used primanly to manage specificafly-targeted aquatic macrophyte species. Herbicides are applied in either a liquid or granular form. Herbicides can be successfully used in most lakes. In those lakes which serve as a potable water supply, however, certain use restrictions may be in place for the herbicides. NYSDEC (1990) lists endothall, diquat, and 2,4-D as the most commonly used aquatic herbicides in NYS. The average cost of most aquatic herbicides ranges between $200 -$400 per treated acre (NYSDEC, 1990). The cost per acre to apply Sonar varies greatly depending on the application rate and the depth of water. In general, the cost may range between $40 - $160 per treated acre

9.2.1 Endothall

Endothall was reviewed by the NYSDEC (1981). Endothall compounds are contact herbicides, which are primarily used for the control of most pondweeds and coontail. Endothall is not effective for floating or emergent species. The active ingredient in endothall is 7-oxabicyclo[2.2.1]heptane-2,3-dicarboxylic acid. The dipotassium salt of endothall is sold under the trade name Aquathol® K, as an aquatic herbicide. The mono(N,N-dimethylalkylamine) salt of endothall is sold under the trade name Hydrothol® 191, as an aquatic algicide and herbicide.

Pullman (1993) notes that the dipotassium salt of endothall will control Eurasian watermilfoil. However, he goes on to note that selective control is not possible because the application rates necessary to control Eurasian watermilfoil are lethal to many native plant species. WSDOE (1992) reports that endothall may have significant adverse impacts on non-target aquatic plants. A treatment concentration of 500 ppb for 72 hours was shown by Netherland et al. (1991) as being an optimum concentration to result in a complete removal of Eurasian watermilfoil in the water column and a shoot biomass reduction of greater than 98% when compared to reference locations.

NYSDEC (1981) notes that endothall is highly toxic to humans. WSDOE lists the acute toxicity of dipotassium or disodium endothall as ranging from 95 ppm for redfin shiners (<u>Notropis</u> <u>umbratilis</u>) to 710 ppm for striped bass (<u>Morone saxatilis</u>) fingerlings. Elf Atochem (1992) reports a tolerance level in water for fish of 60 to 100 ppm of dipotassium or disodium endothall. Toxicity values are significantly lower for the amine formulation of endothall. Endothall is rapidly taken up and produces quick results. This can lead to depleted oxygen levels in the water due to the sudden contribution of decaying plant biomass to the water column. Endothall is neither bloaccumulated nor persistent in the aquatic environment.

Vermont Department of Environmental Conscivation (VDEC, 1993) notes that the advantage of endothall is that it is a fast acting herbicide. They also report that the disadvantages include: 1) the potential need for water use restrictions; 2) the potential need for an alternate water supply for a period of time; 3) the fact that endothall does not kill the roots, only the leaves and stems it comes in contact with; 4) the fact that control is short-termed; and 5) the fact that endothall is not selective for Eurasian watermilfoil.

9.2.2 Diquat

Diquat was reviewed by NYSDEC (1981). Diquat dibromide (6,7-dihydrodipyrido (1,2-a:2’,1’-c)pyrazinediium dibromide) is a contact herbicide that can be selective for Eurasian watermilfoil. Diquat is sold under the tradename Reward®. It is used to control several submergent, floating, and emergent macrophytes at one to two gallons per acre. It is a broad spectrum contact herbicide with only 1ocal plant translocation. It is absorbed through the cuticle and works by interfering with photosynthetic activity within the plant. As a contact herbicide, it is taken up quickly and produces rapid results. This can result in decreased oxygen levels due to the sudden addition of decaying plant biomass to the water column. Pullman (1993) notes that at an application rate of 1 gallon per acre of treatment area, Eurasian watermilfoil will drop out of the water column in 10 days to two weeks, with little impact to aquatic plants native to Michigan. However, Eurasian watermilfoil will rapidly recover from a diquat application. NYSDEC (1981) considers diquat to have moderate toxicity to fish and invertebrates, moderate toxicity to test mammals, high oral toxicity to humans, and moderate to low toxicity to birds.

VDEC (1993) notes that the advantage of diquat is that it is a fast acting herbicide. They also report that the disadvantages include: 1) the potential need for water use restrictions; 2) the potential need for an alternate water supply for a period of time; 3) the fact that diquat does not kill the roots, only the leaves and stems it comes in contact with; 4) that fact that control is short-termed; and 5) the fact that diquat is not selective for Eurasian watermilfoil and water stargrass.

9.2.3 2.4-D

The aquatic herbicide 2,4-D was reviewed by NYSDEC (1981). The active ingredient is a granular formulation of 2,4-dichlorophenoxyacetic acid, butoxyethyl ester. 2,4-D is sold under the tradename Aqua-Kleen®. It is considered to be quite selective for Eurasian watermilfoil. It is a systemic herbicide which kills by inhibiting cellular division, though at low concentrations it may stimulate growth (VDEC, 1993). It is used to control several floating and submerged species, including Eurasian watermilfoil (NYSDEC, 1990). Pullman (1993) reports that when 2,4-D is applied at label-recommended rates, little or no impact to non-target species is observed. NYSDEC (1981) considers 2,4-D to have moderate toxicity to humans, low toxicity to test mammals, low toxicity to birds and varying toxicities to fish. VDEC (1993) reports that a concern has been raised by the USEPA's Office of Pesticide Programs concerning the potential carcinogenicity of 2,4-D, which is being evaluated by that office.

9.3 NON-CHEMICAL ALTERNATIVES

Non-chemical alternatives to Sonar® were evaluated with respect to their effectiveness, their advantages, and their disadvantages. These alternatives could he more suitable for small areas of milfoil or other target aquatic macrophytes (less than five acres for partial treatment) and areas having significant water movement. Generally, the non-chemical alternatives to Sonar® can be divided into mechanical alternatives, biological alternatives, and water level manipulation (drawdowns).

It is important to note that the Vermont Department of Environmental Conservation (VDEC) has been attempting to control the spread of Eurasian watermilfoil through non-chemical means since 1978. The primary mean have been mechanical harvesters and bottom barriers. Despite the attempts at controlling the spread of Eurasian watermilfoil, this aquatic macrophyte has continued to spread within infected lakes where controls have been attempted and to uninfested lakes which had not been targeted for milfoil control measures (VDEC, 1993). The Milfoil Study Committee of the VDEC recommended the use of aquatic herbicides on a site specific basis for the control of introduced, exotic vascular aquatic plant species (VDEC, 1993). The Committee does not recommend the use of Diquat or Endothall because their use would not meet the statutory requirement of pesticide minimization in a long-range management plan and they do not recommend the use of 2,4-D because of the uncertainty about potential human health effects.

9.3.1 Mechanical Alternatives

9.3. 1. 1 Aquatic Weed Harvesters

Harvesters are floating machinery that use a series of blades to cut the aquatic weeds at a point just above the hydrosoil of the water body, depending on depth. Harvesters are effective at removing aquatic vegetation. Madsen et al. (1988b) noted harvesting efficiencies of 79% of Potamogeton pectinatus. Engel (1990) noted that the effectiveness of harvesting is dependent on the time of year it is conducted. In his evaluation, a native macrophyte community harvested in June took a few weeks to reach pre-harvesting biomass. A native macrophyte community harvested in July took until the following spring to reach pre-harvest biomass. In his four year study, Painter (1988) reported that harvesting of a plot in Buckhorn Lake in Ontario in June and September resulted in reduction of Eurasian watermilfoil biomass, shoot weight, and plant density. However, plant height continued to reach the waters surface in the fourth year of the study. Perkins and Sytsma (1987) noted that a single harvest of Eurasian watermilfoil in July produced only a short reduction in the standing crop biomass. A twin harvest program provided an additional 36% reduction in the standing crop biomass. However, in their investigation, Perkins and Sytsma (1987) did not see a long-term reduction in the standing crop as a result of harvesting.

Harvesters have several advantages in that their use results in an immediate reduction in the plant material in the water column. Mechanical harvesters can be used in a limited, confined area and their use generally does not require any type of water use restriction. Another advantage is that they remove the plant biomass from the water. VDEC (1993) notes that the advantages to mechanical harvesting include; 1) mechanical harvesting may be used on a large scale; 2) the method immediately creates open water areas; 3) the fact that the lower part of the plant remains intact to provide some habitat; and, 4) the fact that there is no interference with water supplies or water use.

There are several disadvantages to mechanical harvesting, Because harvesting does not remove the plant roots, regrowth will occur. Generally, the maximum depth that the harvesters blades can reach is approximately six feet. For aquatic species such as Eurasian watermilfoil, growing in excess of six feet of water, a substantial amount of biomass will be uncut. For fast growing species such as Eurasian watermilfoil, regrowth may occur in as little as one month, thereby requiring several harvests during the growing season. Pullman (1993) noted that repeated harvesting during a single growing season has been shown to reduce Eurasian watermilfoil populations. However, because mechanical harvesting is a broad spectrum process, the native plant communities will be as significantly impacted as the target species. The loss of the native plant community can result in the loss of valuable fish and wildlife habitat. Engel (1990) noted that the major ecological impacts of harvesting were changes in the macrophyte community structure and impacts to fish and their invertebrate prey.

Another disadvantage is the production of plant fragments. While harvesters remove most of the cut vegetation from the water column, they are not completely successful. Some plant fragments will be dispersed through the actions of the harvester. For plants such as Eurasian watermilfoil, which spread primarily through the dispersion of plant fragments, this may result in increased aerial coverage of the aquatic weed. Mechanical harvesting will also directly impact fish populations in the treatment area. WSDOE (1992) notes that harvesting can kill up to 25% of small fish in a given treatment area.

Other disadvantages include: 1) the need to have the plants within close proximity of the water surface to facilitate the most efficacious removal; 2) the fact that operating depths are generally limited to five to six feet, with an inability to harvest in shallow water; 3) the need for a disposal site for the harvested plants; 4) the inability to harvest around boats or inside docks; 5) the need for a ramp to launch the harvester; 6) the need for good weather and light winds; and 7) costs that are generally greater than herbicidal control. Harvesters cost between $50,000 and $120,000 per machine and from $200 to $600 per acre to operate for each harvest pass (NYSDEC, 1990 and VDEC, 1993).

9.3.1.2 Benthic Barriers

Benthic barriers are any compound, fabric, or physical structure that can be placed between the sediment and the water column to block sunlight and prevent the photosynthetic activities of the targeted plants. Benthic barriers may drastically alter lake plant and fish communities if used on more than a spot basis. Perkins et al. (1980) have shown that benthic barriers are an effective means of treating Eurasian waterrnilfoil. Eichler et al. (1993) noted that following removal of the benthic barriers, the first species to recolonize the treated areas were native species that overwintered as seeds or turions. In their investigation, Eurasian watermilfoil recolonized 71% of all sites within two years of removal of the barriers, though it was not the dominant species in the community.

The advantages of benthic barriers include multi-year control after initial installation. WSDOE (1992) notes that the effectiveness may range from 1 to 2 years up to 10 years. Benthic barriers can be used in confined areas around docks or in swimming areas. They are generally easy to install and durable, though they can be difficult to install if the water is not shallow. VDEC (1993) notes that the advantages to bottom barriers include: 1) long-term control if properly installed; 2) the method provides immediate control throughout the entire water column; 3) the use in areas not accessible to other mechanical means; and 4) the fact that there is no interference with water supplies or water use if properly installed.

The disadvantages include the high cost of initial installation. NYSDEC (1990) noted that benthic barriers can cost between $2,000 and $8,000 per acre, depending on the choice of fabric. VDEC (1993) considers this technique as not feasible on a large scale because of cost. Benthic barriers often require maintenance on a yearly basis and will require a relatively smooth lake or pond basin substrate. Additionally, benthic barriers may interfere with fish spawning and may significantly impact the benthic invertebrate community (NYSDEC, 1990 and WSDOE, 1992). Bartodziej (1992) noted that the use of benthic barriers in a lake in Florida resulted in significant adverse impacts to the benthic community under the barriers. Further, benthic barriers are not selective within the treatment area.

9.3.1.3 Hand Cutting

Hand cutting or pulling consists of the use of battery operated, knife blade or rake-type implements to cut the target plants. These methods are adequate for control of aquatic weeds inside decks and around boats, along shoreline property and inside swimming areas. This weed management technique is labor intensive, but does not require substantial skill, equipment, or expense (WSDOE, 1992). Bove (1992) utilized this technique in a lake in Vermont and considered the method effective in areas of low Eurasian watermilfoil densities.

VDEC (1993) considers the advantages of this technique to include: I) the selective use in areas of greatest Eurasian watermilfoil density; 2) the potential for use by volunteers to keep costs down; 3) the method can be utilized in rocky and confined areas; 4) the fact that long-term control may be achieved if roots are removed, though fragments from other plants may movc back into the treated area if a whole lake treatment program is not taken; and 5) there is no interference with water supplies or water use. Bove (1992) suggests that volunteers become more difficult to obtain over the course of a long management program, thereby placing a potential labor restraint on this method.

The disadvantages of this alternative include the non-discriminate nature of the method, depending on the type of hand removal. This disadvantage is usually mitigated by the small area of impact. Additional disadvantages include: 1) the fact that plant fragments may be generated which act to spread the target species; 2) the method may result in a short-termed sediment disturbance which would reduce water quality; 3) the fact that a smooth bottom is generally needed; and 4) the fact that the method is too slow and labor intensive to use on a large scale.

9.3.1.4 Rototilling or Rotovating

Rototilling is the use of a hydraulically operated rotovator head from a floating platform that removes the plant roots from the hydrosoil. This method is an effective means of controlling aquatic vegetation (Pullman, 1993). The advantages of this method include the ability to work to a maximum depth of 17 feet. Rototilling allows for seasonal to multiseasonal control of aquatic vegetation, depending on species. Generally, there are no water use restrictions with this method of weed control. It can be performed in a limited area and rototilling can occur over rocks and stumps.

There are several disadvantages to this method. As with mechanical harvesting, this method is broad spectrum and can facilitate the spread of the weed through the generation of plant fragments. Also, because this method occurs in the hydrosoil, a significant sediment load can be generated in the water column which could smother fish eggs arid fry. Invertebrate habitat in the benthic area will be destroyed, which could impact the fish and wildlife species dependent on those organisms. This could result in changes in the aquatic ecosystem. Additionally, faster growing invasive species, such as Eurasian waterrnilfoil, may repopulate the area to the exclusion of slower growing native species (Smith and Barko, 1990; NYSDEC, 1990; and Pullman, 1993). NYSDEC (1990) and VDEC (1993) note that the capital costs for rototilling range from $50,000 to $120,000, with an operating cost of $100 to $1200 per acre.

9.3.1.5 Diver-operated Suction Dredging

This technique consists of the use of suction dredging equipment by scuba-equipped divers to strategically remove the target species. WSDOE (1992) noted that this technique is practical for clearing individual objects such as dock areas or pilings nd can result in up to 90% removal of the desired species. It can be a selective method for either an area or a species (NYSDEC, 1990 and WSDOE, 1992). Eichler eta'. (1991) reported that suction dredging did not eliminate mu foil populations in a single season of harvesting, but was an effective means of managing Eurasian watermilfoil. Bove (1992) noted that diver-operated suction harvesting was used in a lake in Vermont with only limited success, She noted that it was an effective technique in areas of moderate densities of growth. However, it was not effective in dense growth areas as the root systems were difficult to extract from the associated sediments 'arid ecessivc fragmentation of the milfoil was created. Bove also noted that effectiveness varies with bottom sediments type, with rockier sediments being more difficult to remove the plants from than silty sediments.

VIYEC (1993) noted that the advantages to this technique inclHde: 1) the removal of roots; 2) the fact that there is no limitation in water depth to operate; 3) the fact that this method can be selective for Eurasian watermilfoil: 4) the fact that this method can work in areas with underwater obstructions; 5) that control is possible for up to two years; and, 6) the fact that there is no interference with water supplies or water use.

The disadvantages to this method include an increase in turbidity and re-suspension of any contaminants bound in the sediment, decreased water clarity, and a possibility of algal blooms as a result of an increased nutrient load in the water column. Suction dredging will destroy benthic invertebrate habitat, though the effect is generally limited to a small ra2 hecause of the limited nature of the method, VDEC (1993) noted that the disadvantages to this method include:
1) the creation of plant fragments; 2) the necessity for plant disposal; 3) the need for constant machine maintenance; 4) the method is slow and labor intensive; 5) the method is generally applicable for small scale use only; 6) the method disturbs organisms in the benthic zone of a waterbody; 7) the method may result in short-term siltation which would smother fish eggs and fry; and, 8) this method is potentially hazardous to employees due to the necessity for scuba equipment NYSIJEC (1990) estimates that the capital cost of the dredge equipment is about $15,000 to $20,000, with an operating cost of approximately $1,000 to $25,000 per acre.

9.3.2 Biological. Alternatives

Biological methodologies consist of the use of introduced biota to control the targeted aquatic macrophytes. This alternative poses all of the potential problems of the invasive exotic aquatic macrophytes in that once they are released, the biota cannot be controlled. Of the three types of biological alternatives, the use of grass carp (Ctenopharynogodon idella) is not permitted in NYS and the use of insects and plant pathogens are still under study.

To underscore the problems inherent to biological controls, the following is quoted from NYSDEC (1990), Page 6-45:

> "Biological control methods, however, are not well understood. They are relatively new, have not been studied often in the field, and have not been applied to a wide variety of lake conditions. The most significant reason for the tack of understanding about biological controls, however, is in the nature of biological manipulation. Ecosystems are at once dynamic and extremely fragile; a change in one component in the ecosystem can have dramatic effects in other components within the ecosystem. Unlike physical control methods and to a lesser extent, chemical techniques, the results from biological manipulation studies either in theory or in the laboratory cannot be easily reproduced in the field, in actual lakes."

9.3.2.1 Grass Carp

Grass carp are an exotic herbivorous fish that can consume from 20 to 100% of their body weight in vegetation on a daily basis. Generally, only sterile carp are released into waters for vegetation control. NYSUEC (1990) considers that the disadvantages of grass carp use for vegetation control far outweigh their advantages. Unless adequately controlled, fish can escape from the stocked water and move into other waters, where they could impact plant communities in an unwanted fashion. NYSDEC (1990) noted that the most significant disadvantage to the use of grass carp is the potential to completely eradicate aquatic vegetation within a waterbody. This is further exacerbated by the fact that carp will not choose target plants such as Eurasian watermilfoil as their primary diet, instead choosing more native species, such as the pondweeds (NYSDEC, 1990, and Pine and Anderson, 1991). The total removal of the plant community can have extreme consequences to the aquatic ecosystem, significantly affecting native fish, wildlife, vertebrate and invertebrate populations (NYSDEC, 1990). Additionally, parasites have been identified as carried by grass carp. Costs for the use of grass carp range from approximately $50 to $100 per acre.

9.3.2.2 Insects

Various insects have been shown to be effective in controlling aquatic nuisance macrophytes. Generally, these organisms have certain life stages which feed on selected portions of the targeted plants. The larvae of a midge, Cricotopus myriophylli, has been shown to produce significant impacts to Eurasian watermilfoil (Kangasniemi, 1993). Macrae et al. (1990) noted that trials indicated that the larvae are very host-specific to Eurasian watermilfoil. However, more information is needed regarding the extent and specificity of the control. Macrae et al. (1990) noted that the midge only feeds on that portion of the plant extending above the surface of the water, leaving the underwater portion intact. As a controlling agent then, this alternative would not address the issue of Eurasian watermilfoil in a waterbody. NYSDEC (1990) noted that most of the successful applications of insects as a controlling agent have occurred in the southern United States. NYSDEC (1990) goes on to note that insects have been used effectively in conjunction with short-term control programs such as herbicidal or mechanical treatment, to produce long-term control. There is no indication as to the projected cost of this alternative

9.3.2.3 Pathogens

Pathogens are biological agents that produce disease and death in. the targeted organism. Pullman (1993) noted that a fungal, pathogen, Mycoleptodiscus terrestris, has been shown to be a possible biological agent for the control and management of Eurasian watermilfoil. Much of the research has been conducted through the U.S. Army Corps of Engineers Waterways Experiment station in Vicksburg, Mississippi. This technique is currently a research project and pathogens are not available for use on Eurasian watermilfoil or other submersed northern species. There is no indication of the potential cost for this alternative.

9.3.3 Water Manipulation - Drawdown

Drawdowns or water level control is an activity in which the level of the lake is lowered to expose aquatic vegetation in shallow nearshore areas to the elements with the aim to eradicate it. Drawdowns are usually limited to those lakes or ponds which have a dam structure or similar mechanism for controlling the level of water. NYSDEC (1990) noted that the only beneficial time for a drawdown is in winter. NYSDEC (1990) goes on to note that for a drawdown to have a significant effect, the water level must he lowered at least three feet, the plants must be exposed for at least four weeks, and the bottom sediments must be frozen to a depth of at least four inches. Article 15, Title 8 of the Environmental Conservation Law presents the regulations associated with the volume, timing, and rate of change of reservoir releases.

Jenkins (1989) noted that a drawdown conducted at Lake Bomoseen in Vermont resulted in a 60% reduction of cover by aquatic species and a 99% reduction in cover by floating aquatic species. Local diversity was reduced by 44%. However, the abundance of a legally protected species was reduced by 86% and a rare species proposed for legal protection was completely removed from the lake. Additionally, he reported that the drawdown damaged the lake bottom, producing nutrient releases. VDEC (1990) noted that Eurasian watermilfoil was reduced in exposed areas of Lake Bomoseen; however, because it was not impacted in the deeper sections of the lake, recolonization of the shallower sections was expected.

VDEC (1993) considers the advantage of this technique to be the low operational cost and the potential for longer-term control than with other methods, though this would only be the situation if the whole benthic zone was exposed. Impacts to aquatic macrophytes from drawdowns are mixed, depending on species. Drawdowns have been shown to affect fanwort, coontail, most species of milfoil, most species of yellow waterlilies, and bladderwort. Drawdowns have been shown to have little effect on Chara spp., elodea, cattails, and tapegrass (Vallisneria americana). Drawdowns have been shown to increase the populations of most species of pondweeds (NYSDEC, 1990).

Disadvantages include the possible depletion of oxygen in the remaining water, if the lake is shallow and there is a high oxygen demand in the sediments and stream inflow. This could possibly result in fish kills. A nutrient release could result upon restoring the original water levels, which can produce algal blooms. Other macrophyte species may emerge as a result of the drawdown. Increased turbidity and resuspension of sediments may occur (NYSDEC, 1990). VDEC (1993) lists the disadvantages of this technique as being: 1) the potential for significant impact to non-target plants, invertebrates, fish and wildlife; 2.) the potential for impacts to water intakes and shallow wells; and, 3) method effectiveness and lake refill depends on the weather.

9.4 INTEGRATED PEST MANAGEMENT

The optimal method of addressing aquatic macrophyte concerns is in a coordinated effort that brings the most effective and environmentally sound techniques to bear on the problem. An integrated approach would be based on the use of all techniques, depending on the characteristics of the specific problem in a waterbody. An integrated approach, however, would not only be based on a variety of techniques to address the immediate issue of excessive aquatic macrophyte growth, but also the inherent causes of the problem. Such an approach would include measurcs to reduce artificially stimulated lake eutrophication that exacerbates nuisance weed growth. Such activities would include measures such as management and control of nutrient loading, reduction of wastewater flow and reduction of sedimentation on a lake watershed basis. However, such techniques can be expensive and slow to implement. Integrated pest management is an ideal goal of lake management, but is not always a practical solution. A detailed discussion of Integrated Pest Management is presented in Diet For a Small Lake (NYSDEC, 1990).

9.5 ALTERNATIVES ANALYSIS

As discussed throughout Sections 2.0 and 3.0 of this GEIS, the uncontrolled growth of aquatic rnacrophytes in surface waterbodies can substantially impact the ecological characteristics of that waterbody. Desired water uses such as recreational uses may also be prevented or made hazardous by unwanted plant growth. This is particularly true for exotic species such as Eurasian watermilfoil and curlyleaf pondweed, which are capable of exponential growth. It is the responsibility of the lake manager or lake association to decide upon a course of action that not only effectively controls the macrophyte of concern, but also is ecologically sound. The use of the aquatic herbicide Sonar® is one of the alternatives that is available for the control of aquatic macrophytes. This section describes a general approach to deciding upon the use of Sonar® with respect to the other alternatives described in Section 9.

It is the responsibility of the lake manager or lake association to monitor their lakes or ponds with respect to its plant populations, including the growth and distribution of exotic and indigenous macrophytes. Through these monitoring efforts, the infestation of the waterbody by exotic macrophytes or the excessive growth of macrophytes would be noted. Any subsequent decisions regarding macrophyte management approaches must consider all permit requirements, including those specified in Part 327 as described in Section 7.4.

To document the infestation, particularly in advance of a Part 327 permit application, information on the nature and extent of the inestation would be required. That information would include the nature and areal coverage of the infestation, the areal size of the waterbody, the location of the infestatopm with respect to the watcrbody, the depth of the water column, the recreational uses of the waterbody, the location and distances of potable water intakes with respect to the potential treatment zone, other macrophyte species which may be present, and the presence and distribution of any rare species. Information on sediment types and water movements should also be gathered. Other important considerations would be the lake management objectives and any criteria under the NYS Freshwater Wetlands Act.

Much of this information is available directly off of maps and diagrams produced by the NYSDEC. The nature of the macrophytes in and surrounding the infestation area can be determined through either direct visual observation (non-harvesting methods) or by clipping samples of the littoral vegetation for identification (harvesting methods). Community characteristics such as horizontal and vertical zonation, plus frequency and dominance can be determined by the collection of a number of samples in relationship to the area of concern. The depth of the water column can either be determined through electronic means (Sonar) or through mechanical means (drop-lines and staff gauges).

As noted in Section 3.0, small quantities of Eurasian watermilfoil and curlyleaf pondweed in the early stages of infestation may offer many of the functions and values of native aquatic macrophytes In this instance, the no-action alternative may be an appropriate management strategy. The lake manager or Lake association would monitor the growth patterns of the areas of infestation under such a strategy. If the infestation is highly localized, the lake manager or lake association may chose a technique such as hand pulling, benthic barriers, or suction dredging as a control option. If the decision by the lake manager or lake association is that the quantity of macrophytes in the waterbody of concern is posing an ecological, recreational, or safety impact to the use of the waterbody, an appropriate management approach may be chosen using the following guidelines.

In ponds less than five acres in size where the entire waterhody is sbustantially dominated by macrophytes targeted for control, Sonar® would be an effective control method, particularly with respect to Eurasian watermilfoil and curlyleaf pondweed. In comparison to the other possible herbicides, neither Endothall nor Diquat are selective for Eurasian watermilfoil. 2,4-D is selective for Eurasian watermilfoil, but has greater water use restrictions than fluridone. Other herbicides may not be selective to control only targeted species. With respect to mechanical alternatives, Sonar® would produce longer lasting results with less environmental damage than mechanical harvesting, benthic barriers or dredging. Drawdown also is not a preferred option as it is not always a choice with a particular waterbody and the drawdown may not be able to effect the deeper parts of the pond. The potential ecological impacts from drawdowns include the possible depletion of oxygen in the remaining water, which could result in fish kills, and nutrient releases, which could produce algal blooms and increase the spread of other macrophyte species. Increased turbidity and resuspension of sediments may occur (NYSDEC, 1990). Other disadvantages are listed in Section 9.3.3.

Within a larger lakes, if the area to be treated is less than 5 acres in size, a contact herbicide such as Endothall or Diquat may be an appropriate control method. A systemic herbicide such as 2,4-D may also prove effective, if water use restriction can be met. The Sonar® label states that treating areas less than five acres in size may not produce satisfactory results due to dilution by untreated water. Mechanical alternatives such as benthic barriers or raking would also be possible treatment choices, and woutd be more cost effective than harvesting.

Where the area to be treated is greater than five acres, Sonar® would be an appropriate alternative. In comparison to the other possible herbicides, Endothall and Diquat are non-selective for Eurasian watermilfoil and do not provide long-term control of Eurasian watermilfoil. 2,4-D is selective for Eurasian watermilfoil, but has stricter water use restrictions than fluridone. With respect to mechanical alternatives, Sonar® would produce longer lasting results, with less environmental damage, than mechanical harvesting. VDEC (1993) notes that there are significant environmental impacts associated with the use of mechanical alternatives. Drawdown often is not a choice with a particular waterbody and the drawdown may not be able to effect the deeper parts of the lake. The potential ecological impacts from drawdowns include: possible depletion of oxygen in the remaining water that could result in fish kills; and nutrient releases which could produce algal blooms and increase the spread of other macrophyte species. Increased turbidity and resuspension of sediments may occur (NYSDEC, 1990). Other disadvantages are listed in Section 9.3.3.

As discussed in Section 9.3.2, biological alteatives in NYS are either not permitted or are still in the testing phase. At present, biological alternatives are not developed for use.

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Supplemental Environmental Impact Statement
Assessments of Aquatic Herbicides

WASHINGTON STATE
DEPARTMENT OF
ECOLOGY

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Washington State Department of Ecology
Water Quality Program

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APPENDICES

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:	Untied 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
Contract Authors and Principle Contributors:	Bernalyn D. McGaughey, Risk Assessment Project Manager Compliance Services International 1112 Alexander Avenue E. Tacoma, Washington 98421 Robert G. Dykeman, MS University of Washington (Fisheries Science, minor Aquatic Biology); BS University of Washington (Fisheries, minor Limnology, Freshwater Biology), AS Tacoma community college (Sciences).

	Jon E. Ford, Ph.D., Ph.D. Wayne State University, School of Medicine, (Psychology and Pharmacology); MS Wayne State University, School of Medicine, (Occupational and Environmental Health); BS Lewis and Clark College, (Biology and Psychology). Stephen O. Jacobson, BA University of Washington, (Chemistry). Richard P. Kemman, MS Clemson University, (Environmental Toxicology); BA Whitworth College, (Biology). Thomas M. Priester, Ph.D. University of California, Riverdale (Entomology, Aquatic Biology and Medical Entomology). BS University of California, Irvine (Aquatic Biology and Molecular Biology).
Agency Authors and Principle Contributors:	Washington State Department of Ecology P.O. Box 47600 Olympia, WA 98504-7600 Kathleen Emmett, MPA The Evergreen State College (Political Administration), BSED Eastern Montana College (English, Political Science and Education). Kathleen S. Hamel, MS University of Western Australia (Aquatic Ecology), BA the University of Washington. Joan Hardy, Ph.D. University of Washington, MS University of Washington, BA Whitman College (Biology, Limnology, Aquatic Toxicology). Kari Rokstad, AA Green River Community College. Stephen L. Saunders, MS The Evergreen State College (Environmental Studies), BS Western Washington University (Huxley College of Environmental Studies).
Issue Date of Draft SEIS:	July 10, 2000
Date Comments on Draft are Due:	August 11, 2000
Projected Date of Final SEIS:	September 2000
Location of Background Data:	Water Quality Program Department of Ecology P.O. Box 47600 Olympia, WA 98504-7600

For copies of the SEIS documents: Please contact Kathleen Emmett, (360) 407-6478, or email kemm461@ecy.wa.gov or write to the above Ecology address.

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 Untied 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 be 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 you 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 knapwood, 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 ghaubrich@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 staff's 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 rotovation, 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, rotovation 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 rotovation 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 it 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, re-stocking, 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 rotovation 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 rotovation. 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 rotovation 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 rotovation 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, permitters 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, permitters 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. Permitters 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. The 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 ecosites 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 meristimatic 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 (*Euhrychiopsis 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 effort 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^{o} 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^{o} F, while feeding diminishes below 53^{o} 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^{o} F but below 90^{o} F. Grass carp can survive a wide range of temperatures from freezing to 95^{o} 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 favorable 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, popnets, 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 usiemens, 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 $Fe(OH)_3$ 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 that 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 *Potamogton* > *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^{o} 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 sibericum*, 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 though 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.
- Systematic 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 eco-tox 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 (¼) 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 (¼) 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 (½) 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 (¼) mile and one-half (½) mile application restriction for potable water use [i.e., water treated with glyphosate (Rodeo®) should not be used for drinking water within one-half (½) 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 (¼) 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. <u>Posting on the Water</u>:

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 Sate 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-Dicholorophenoxy 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 Dimethlyamine 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 CO_2. 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 CO_2 (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-chorolpehnol which are both very toxic to some species of aquatic macrophytes, marine phaeophytes, various beneficial fungal species and amphipods. Failure to control emersed, 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 ingests 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 tropic 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 carbons 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 anadromonous 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 *Limondynastes 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 algaecide), 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, <u>Review of the Drinking Water Criteria Document for Copper</u>. 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 LC_{50} or EC_{50}) 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 (<u>Oncorhynchus</u> <u>mykiss</u>, formerly <u>Salmo</u> <u>gairdneri</u>) 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 (<u>Salmo</u> <u>trutta</u>), and Atlantic salmon (<u>Salmo</u> <u>salar</u>). 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 (<u>Oncorhynchus tshawytscha</u>) 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 (<u>Salvelinus</u> <u>fontinales</u>), 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 (<u>O</u>. <u>mykiss</u>) 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 (<u>O</u>. <u>kisutch</u>) 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 LC_{50} for rainbow trout was 0.8 mg/l copper (Herbert et al. 1965). Brown (1968) also estimated 48-hour LC_{50} 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 LC_{50} 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 LC_{50} of 1.1 mg/l for rainbow trout, also with hard water (320 mg/l as $CaCO_3$). With soft water (15-20 mg/l as $CaCO_3$) the 72-hour LC_{50} for rainbow trout was 0.44 mg/l. In another study, investigators found a 96-hr LC_{50} in hard water (290-310 mg/l as $CaCO_3$) 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 LC_{10} (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 $CaCO_3$), Goettleet 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 vulgans*) (Westerdahl and Getsinger 1988). A BCF of 290 was measured for the fathead minnow (P*imephales 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 LC_{50} for rainbow trout was 3.6 ppm; an LC_{50} 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 (LC_{50} = 1.5 ppm) to rainbow trout, while at 12 degrees centigrade, copper sulfate became highly toxic (LC_{50}=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 $CaCO^3$) 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 (CaC03), 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 dipyridylium 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 submersed 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 - LC_{50}
  2. Freshwater fish - LC_{50}
  3. Aquatic LC_{50} (invertebrates)
  4. Aquatic LC_{50}
  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 LD_{50} (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 abraised 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 (*Utricularia* spp.) Naid (*Najas* spp.) Coontail (*Ceratophyllum demersum*) Watermilfoil (*Myriophyllum* spp.) Pondweed (*Potamogeton* spp. except *P. robbinsii*) Elodea (*Elodea* spp.) Hydrilla (*Hydrilla verticillata*)
Floating weeds	Waterhyacinth (*Eichhornia crassipes*) Salvinia (*Salvinia rotundifolia*) Waterlettuce (*Pistia stratiotes*) Pennywort (*Hydrocotyle umbellata*) Duckweed (*Lemna* spp.)
Marginal weeds	Cattail (*Typha* spp.)
Algae	*Pithophora* spp. *Spirogyra* 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 Hyalella 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 LD_{50} in twelve young male mallard ducks was 564 mg/kg (Hudson 1984). The LC_{50} 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 restrict 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 <u>and</u> 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 CO_2, 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 bioaccumlate, 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 hydrellia fly. At concentrations of Aquathol® K that controlled Hydrilla, 74% of hydrellia 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.), spadderdock (*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 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 use 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 CO_2, 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).

Water

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 carbons 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 bioaccumlate 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 vulgarisis*). 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 ppm a.e./ 0.079ppm a.e. = ~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 ppm a.e./ 0.080 ppm a.e. = ~ 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 endothall 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 Aqauthol® K, disodium endothall salt or endothall 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 consentrations (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 emersed 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 hydrosoils 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 hydrosoil 14 days after treatment (Grant et al. 1979). No detectable residue (at a test sensitivity of 0.010 ppm in hydrosoil) was observed in hydrosoil 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 hydrosoil 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 hydrosoil 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 hydrosoil. Additionally, laboratory conditions minimized hydrosoil 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 hydrosoils remains a concern. Since fluridone can be absorbed from the hydrosoil by roots, and since fluridone persists in the hydrosoil 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	*Hydrilla verticillata*
American Elodea	*Elodea canadensis*
Fanwort	*Cabomba caroliniana*
Eurasian watermilfoil	*Myriophyllum spicatum*
Coontail	*Ceratophyllum demersum*
Illinois pondweed	*Potamogeton illinoensis*
Southern naiad	*Najas guadalupensis*
Parrotfeather	*Myriophyllum brasiliensis*
Common bladderwort	*Utricularia spp.*
Water celery	*Vallisneria spp.*
Arrowhead	*Sagittaria* spp.
Cattail	*Typha* spp.
Spatterdock	*Nuphar luteum*
Reed canarygrass	*Phalaris arundinacea*

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 ug/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	
Daphnia	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 looses 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 emersed aquatic grasses, broadleaf weeds, and brush (Westerdahl and Getsinger 1988). Glyphosate is applied to emersed 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 $^{14}CO_2$ and the subsequent photofixation of respired $^{14}CO_2$ 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 emersed 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)
(Chironomus plumosus)	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 LC_{50} 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 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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Fluridone Ecological Risk Assessment, Final Report

ENSR International

Bureau of Land Management

Reno, Nevada

Fluridone
Ecological Risk Assessment

Final Report

November 2005

Bureau of Land Management Contract No. NAD010156
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Executive Summary

The Bureau of Land Management (BLM), United States Department of the Interior (USDI), is proposing a program to treat vegetation on up to six million acres of public lands annually in 17 western states in the continental United States (U.S.) and Alaska. As part of this program, the BLM is proposing the use of ten herbicide active ingredients (a.i.) to control invasive plants and noxious weeds on approximately one million of the 6 million acres proposed for treatment. The BLM and its contractor, ENSR, are preparing a Vegetation Treatments Programmatic Environmental Impact Statement (EIS) to evaluate this and other proposed vegetation treatment methods and alternatives on lands managed by the BLM in the western continental US and Alaska. In support of the EIS, this Ecological Risk Assessment (ERA) evaluates the potential risks to the environment that would result from the use of the herbicide fluridone, including risks to rare, threatened, and endangered (RTE) plant and animal species.

One of the BLM's highest priorities is to promote ecosystem health, and one of the greatest obstacles to achieving this goal is the rapid expansion of invasive plants (including noxious weeds and other plants not native to the region) across public lands. These invasive plants can dominate and often cause permanent damage to natural plant communities. If not eradicated or controlled, invasive plants will jeopardize the health of public lands and the activities that occur on them. Herbicides are one method employed by the BLM to control these plants.

Herbicide Description

Fluridone is a selective systemic herbicide that inhibits carotene production in leaves, which causes the breakdown of chlorophyll—preventing the plant from synthesizing food. This herbicide comes in two formulations: liquid and pellet. Fluridone is used by the BLM for vegetation control in their Aquatic program. Application is carried out through both aerial and ground dispersal. Aerial dispersal is executed through the use of a plane or helicopter. Ground applications are executed on foot or horseback with backpack sprayers, or from all terrain vehicles or trucks equipped with spot or boom/broadcast sprayers. The BLM applies fluridone at different rates depending on the waterbody category (i.e., Ponds, Whole Lake/Reservoir, Partial Lakes/Reservoir, or Canals). In order for the risk assessment simulations to span the concentration range of applied herbicide in typical and maximum cases, the lowest typical application rate (Whole Lake/Reservoir) was selected for use as the typical rate and the highest maximum application rate (Partial Lake/Reservoir) was selected for use as the maximum application rate. The lowest typical application of fluridone is 0.15 pounds (lbs) a.i. per acre (a.i./ac). The maximum application rate is 1.3 lbs a.i./ac.

Ecological Risk Assessment Guildelines

The main objectives of this ERA were to evaluate the potential ecological risks from fluridone to the health and welfare of plants and animals and their habitats and to provide risk managers with a range of generic risk estimates that vary as a function of site conditions. The categories and guidelines listed below were designed to help the BLM determine which of the proposed alternatives evaluated in the EIS should be used on BLM-managed lands.

- Exposure pathway evaluation – The effects of fluridone on several ecological receptor groups (i.e., terrestrial animals, non-target terrestrial plants, fish and aquatic invertebrates, and non-target aquatic plants) via particular exposure pathways were evaluated. The resulting exposure scenarios included the following:
    - direct contact with the herbicide or a contaminated waterbody;
    - indirect contact with contaminated foliage;
    - ingestion of contaminated food items;
    - off-site drift of spray to terrestrial areas; and
    - accidental spills to waterbodies.

- Definition of data evaluated in the ERA – Herbicide concentrations used in the ERA were based on typical and maximum application rates provided by the BLM. These application rates were used to predict herbicide concentrations in various environmental media (e.g., soils, water). Some of these calculations required the computer model AgDRIFT®, which was used to estimate off-site herbicide transport due to spray drift, and an additional sensitivity model designed to determine how pond and stream volumes affect exposure concentrations

- Identification of risk characterization endpoints – Endpoints used in the ERA included acute mortality; adverse direct effects on growth, reproduction, or other ecologically important sublethal processes; and adverse indirect effects on the survival, growth, or reproduction of salmonid fish. Each of these endpoints was associated with measures of effect such as the no observable adverse effect level (NOAEL) and the median lethal effect dose and median lethal concentration (LD_{50} and LC_{50}).

- Development of a conceptual model – The purpose of the conceptual model is to display working hypotheses about how fluridone might pose hazards to ecosystems and ecological receptors. This is shown via a diagram of the possible exposure pathways and the receptors evaluated for each exposure pathway.

In the analysis phase of the ERA, estimated exposure concentrations (EECs) were identified for the various receptor groups in each of the applicable exposure scenarios via exposure modeling. Risk quotients (RQs) were then calculated by dividing the EECs by herbicide- and receptor-specific or exposure media-specific Toxicity Reference Values (TRVs) selected from the available literature. These RQs were compared to Levels of Concern (LOCs) established by the United States Environmental Protection Agency (USEPA) Office of Pesticide Programs (OPP) for specific risk presumption categories (i.e., acute high risk, acute high risk potentially mitigated through restricted use, acute high risk to endangered species, and chronic high risk).

Uncertainty

Uncertainty is introduced into the herbicide ERA through the selection of surrogates to represent a broad range of species on BLM-managed lands, the use of mixtures of fluridone with other herbicides (tank mixtures) or other potentially toxic ingredients (i.e., degradates, inert ingredients, and adjuvants), and the estimation of effects via exposure concentration models. The uncertainty inherent in screening level ERAs is especially problematic for the evaluation of risks to RTE species, which are afforded higher levels of protection through government regulations and policies. To attempt to minimize the chances of underestimating risk to RTE and other species, the lowest toxicity levels found in the literature were selected as TRVs; uncertainty factors were incorporated into these TRVs; allometric scaling was used to develop dose values; model assumptions were designed to conservatively estimate herbicide exposure; and indirect as well as direct effects on species of concern were evaluated.

Herbicide Effects

Literature Review

According to the Ecological Incident Information System (EIIS) database run by the USEPA OPP, fluridone has been associated with only one reported “ecological incident” involving damage or mortality to non-target flora. It was listed as probable that direct contact of fluridone was responsible.

A review of the available ecotoxicological literature was conducted in order to evaluate the potential for fluridone to negatively directly or indirectly affect non-target taxa. This review was also used to identify or derive TRVs for use in the ERA. The sources identified in this review indicate that fluridone has low toxicity to most terrestrial species. Studies conducted with mammals found that acute exposure to fluridone does not commonly cause adverse effects, even to mammals that were exposed to fluridone for longer periods of time or during pregnancy. Similarly, short-term exposure to fluridone did not result in adverse effects in birds, even at high exposure levels. Long-term exposure to fluridone did result in reduced growth in large and small birds. Fluridone was practically non-toxic to honeybees

(*Apis* spp.). While no quantitative data were found to evaluate fluridone's effects on terrestrial plants, qualitative results indicate that the sensitivity of terrestrial plants is variable. Some plant species (e.g., grasses and sedges) were more sensitive than others (e.g., willow).

Fluridone is an herbicide used to control aquatic plants. In the available literature, aquatic plants were not affected by concentrations up to 1 milligrams (mg) a.i./liter (L) (typical herbicide application rates used in the direct spray scenarios in this ERA resulted in a pond concentration of 0.017 mg a.i./L and a stream concentration of 0.084 mg a.i./L). Acute and chronic toxicity tests indicate that fluridone causes toxicity to fish species at concentrations of 10 mg/L, with some adverse effect concentrations approaching 1 mg/L. Acute toxicity concentrations for aquatic invertebrates reached 1.3 mg/L. No data were found to evaluate the toxicity of fluridone to amphibians.

Ecological Risk Assessment Results

Based on the ERA conducted for fluridone, there is the potential for risk to selected ecological receptors from exposure to herbicides under specific conditions on BLM-managed lands. The following bullets summarize the risk assessment findings for fluridone under each evaluated exposure scenario:

- Direct Spray – No risks were predicted for terrestrial wildlife (i.e., insects, birds, or mammals). Risks to terrestrial plants could not be evaluated as a result of a lack of toxicity information; however, one ecological incident report suggests the potential for risk to terrestrial plants. No risks to non-target aquatic plants are predicted when waterbodies are accidentally (streams) or intentionally (ponds) sprayed, but risks to fish or aquatic invertebrates may occur when waterbodies are accidentally or intentionally sprayed.

- Off-Site Drift to Non-Target Terrestrial Plants – Risks to terrestrial plants could not be evaluated because of a lack of toxicity information; however, product literature and one ecological incident report suggest the potential for risk.

- Accidental Spill to Pond – Risk to fish, aquatic invertebrates, and non-target aquatic plants may occur when herbicides are spilled directly into the pond.

Based on the results of the ERA, it is unlikely that RTE species would be harmed by appropriate use (see following section) of the herbicide fluridone on BLM-managed lands.

Recommendations

The following recommendations are designed to reduce potential unintended impacts to the environment from the application of fluridone:

- Select adjuvants carefully (none are currently ingredients in fluridone-containing Sonar products) since these have the potential to increase the level of toxicity above that predicted for the a.i. alone. This is especially important for application scenarios that already predict potential risk from the a.i. itself.

- Review, understand, and conform to "Environmental Hazards" section on herbicide label. This section warns of known pesticide risks to wildlife receptors or to the environment and provides practical ways to avoid harm to organisms or the environment.

- Avoid accidental direct spray on the stream to reduce the most significant potential impacts.

- Because the effects of normal herbicide application on terrestrial plants are uncertain, limit fluridone use in areas where RTE plants are near application areas. Avoid accidental direct spray and off-site drift to terrestrial plants to reduce potential impacts observed in a previous ecological incident report (Section 2.3). Limit fluridone application in wind, and monitor effects on adjacent terrestrial vegetation.

- Use the typical application rate in the pond to reduce risk to fish and aquatic invertebrates.

The results from this ERA assist the evaluation of proposed alternatives in the EIS and contribute to the development of a Biological Assessment (BA), specifically addressing the potential impacts to proposed and listed RTE species on western BLM treatment lands. Furthermore, this ERA will inform BLM field offices on the proper application of fluridone to ensure that impacts to plants and animals and their habitat are minimized to the extent practical.

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LIST OF ACRONYMS, ABBREVIATIONS, AND SYMBOLS

ac	-	acres
a.i.	-	active ingredient
BA	-	Biological Assessment
BCF	-	Bioconcentration Factor
BLM	-	Bureau of Land Management
BO	-	Biological Opinion
BW	-	Body Weight
°C	-	Degrees Celsius
CBI	-	Confidential Business Information
cm	-	centimeter
cms	-	cubic meters per second
CWE	-	Cumulative Watershed Effect
DPR	-	Department of Pesticide Registration
EC_{25}	-	Concentration causing 25% inhibition of a process (Effect Concentration)
EC_{50}	-	Concentration causing 50% inhibition of a process (Median Effective Concentration)
EEC	-	Estimated Exposure Concentration
EIS	-	Environmental Impact Statement
EIIS	-	Ecological Incident Information System
EFED	-	Environmental Fate and Effects Division
ERA	-	Ecological Risk Assessment
ESA	-	Endangered Species Act
FIFRA	-	Federal Insecticide, Fungicide, and Rodenticide Act
FOIA	-	Freedom of Information Act
ft	-	feet
g	-	grams
gal	-	gallon
GLEAMS	-	Groundwater Loading Effects of Agricultural Management Systems
HHRA	-	Human Health Risk Assessment
HSDB	-	Hazardous Substances Data Bank
IPM	-	Integrated Pest Management
IRIS	-	Integrated Risk Information System
ISO	-	International Organization for Standardization
IUPAC	-	International Union of Pure and Applied Chemistry
Kd	-	Partition coefficient
kg	-	Kilogram
K_{oc}	-	Organic carbon-water partition coefficient
K_{ow}	-	Octanol-water partition coefficient
L	-	Liters
lb(s)	-	pound(s)
LC_{50}	-	Concentration causing 50% mortality (Median Lethal Concentration)
LD_{50}	-	Dose causing 50% mortality (Median Lethal Dose)
LOAEL	-	Lowest Observed Adverse Effect Level
LOC(s)	-	Level(s) of Concern
Log	-	Common logarithm (base 10)
m	-	meters
mg	-	milligrams

LIST OF ACRONYMS, ABBREVIATIONS, AND SYMBOLS (Cont.)

mg/kg	-	milligrams per kilogram
mg/L	-	milligrams per liter
mmHg	-	millimeters of mercury
MSDS	-	Material Safety Data Sheet
MW	-	Molecular Weight
NMFS	-	National Marine Fisheries Service
NOAA	-	National Oceanic and Atmospheric Administration
NOAEL	-	No Observed Adverse Effect Level
OPP	-	Office of Pesticide Programs
OPPTS	-	Office of Pollution Prevention and Toxic Substances
ORNL	-	Oak Ridge National Laboratory
ppm	-	parts per million
RQ	-	Risk Quotient
RTE	-	Rare, Threatened, and Endangered
RTEC	-	Registry of Toxic Effects of Chemical Substances
SDTF	-	Spray Drift Task Force
TOXNET	-	National Library of Medicines Toxicology Data Network
TP	-	Transformation Product
TRV	-	Toxicity Reference Value
TSCA		Toxic Substances Control Act
US	-	United States
USDA	-	United States Department of Agriculture
USDI	-	United States Department of Interior
USEPA	-	United States Environmental Protection Agency
USFWS	-	United States Fish and Wildlife Service
µg	-	micrograms
>	-	greater than
<	-	less than
=	-	equal to

1.0 INTRODUCTION

The Bureau of Land Management (BLM), United States Department of the Interior (USDI), is proposing a program to treat vegetation on up to six million acres of public lands annually in 17 western states in the continental United States (U.S.) and Alaska. The primary objectives of the proposed program include fuels management, weed control, and fish and wildlife habitat restoration. Vegetation would be managed using five primary vegetation treatment methods - mechanical, manual, biological, chemical, and prescribed fire.

The BLM and its contractor, ENSR, are preparing a *Vegetation Treatments Programmatic Environmental Impact Statement* (EIS) to evaluate proposed vegetation treatment methods and alternatives on lands managed by the BLM in the western continental US and Alaska (ENSR 2004a). As part of the EIS, several ERAs and a Human Health Risk Assessment (HHRA; ENSR 2004b) were conducted on several herbicides used, or proposed for use, by the BLM. These risk assessments evaluate potential risks to the environment and human health from exposure to these herbicides both during and after treatment of public lands. For the ERAs, the herbicide a.i. evaluated were tebuthiuron, diuron, bromacil, chlorsulfuron, sulfometuron-methyl, diflufenzopyr, Overdrive® (a mix of dicamba and diflufenzopyr), imazapic, diquat, and fluridone. The HHRA evaluated the risks to humans from only six a.i. (sulfometuron-methyl, imazapic, diflufenzopyr, dicamba, diquat, and fluridone) because the other a.i. were already quantitatively evaluated in previous EISs (e.g., BLM 1991). [Note that in the HHRA, Overdrive® was evaluated as its two separate components, dicamba and diflufenzopyr, as these two a.i. have different toxicological endpoints, indicating that their effects on human health are not additive.] The purpose of this document is to summarize results of the ERA for the herbicide fluridone.

Updated risk assessment methods were developed for both the HHRA and ERA and are described in a separate document, *Vegetation Treatments Programmatic EIS Ecological Risk Assessment Methodology* (hereafter referred to as the "Methods Document;" ENSR 2004c). The methods document provides, in detail, specific information and assumptions used this ERA.

1.1 Objectives of the Ecological Risk Assessment

The purpose of the ERA is to evaluate the ecological risks of ten herbicides on the health and welfare of plants and animals and their habitats, including threatened and endangered species. This analysis will be used by the BLM, in conjunction with analyses of other treatment effects on plants and animals, and effects of treatments on other resources, to determine which of the proposed treatment alternatives evaluated in the EIS should be used by the BLM. The BLM Field Offices will also utilize this ERA for guidance on the proper application of herbicides to ensure that impacts to plants and animals are minimized to the extent practical when treating vegetation. The US Fish and Wildlife Service (USFWS) and National Oceanic and Atmospheric Administration Fisheries Service (NOAA Fisheries), in their preparation of a Biological Opinion (BO), will also use the information provided by the ERA to assess the potential impact of vegetation treatment actions on fish and wildlife and their critical habitats.

This ERA, which provides specific information regarding the use of the terrestrial herbicide fluridone, contains the following sections:

Section 1: Introduction

Section 2: BLM Herbicide Program Description – This section contains information regarding herbicide formulation, mode of action, and specific BLM herbicide use, which includes application rates and methods of dispersal. This section also contains a summary of incident reports documented with the United States Environmental Protection Agency (USEPA).

Section 3: Herbicide Toxicology, Physical-Chemical Properties, and Environmental Fate – This section contains a summary of scientific literature pertaining to the toxicology and environmental fate of fluridone in terrestrial and aquatic environments, and discusses how its physical-chemical properties are used in the risk assessment.

Section 4: Ecological Risk Assessment – This section describes the exposure pathways and scenarios and the assessment endpoints, including potential measured effects. It provides quantitative estimates of risks for several risk pathways and receptors.

Section 5: Sensitivity Analysis – This section describes the sensitivity of each of three models used for the ERA to specific input parameters. The importance of these conditions to exposure concentration estimates is discussed.

Section 6: Rare, Threatened, and Endangered Species (RTE) – This section identifies RTE species potentially directly and/or indirectly affected by the herbicide program. It also describes how the ERA can be used to evaluate potential risks to RTE species.

Section 7: Uncertainty in the Ecological Risk Assessment – This section describes data gaps and assumptions made during the risk assessment process and how uncertainty should be considered in interpreting results.

Section 8: Summary – This section provides a synopsis of the ecological receptor groups, application rates, and modes of exposure. This section also provides a summary of the factors that most influence exposure concentrations with general recommendations for risk reduction.

2.0 BLM HERBICIDE PROGRAM DESCRIPTION

2.1 Problem Description

One of the BLM's highest priorities is to promote ecosystem health, and one of the greatest obstacles to achieving this goal is the rapid expansion of weeds across public lands. These invasive plants can dominate and often cause permanent damage to natural plant communities. If not eradicated or controlled, noxious weeds will jeopardize the health of public lands and the myriad of activities that occur on them. The BLM's ability to respond effectively to the challenge of noxious weeds depends on the adequacy of the agency's resources.

Millions of acres of once healthy, productive rangelands, forestlands and riparian areas have been overrun by noxious or invasive weeds. Noxious weeds are any plant designated by a federal, state, or county government as injurious to public health, agriculture, recreation, wildlife, or property (Sheley et al. 1999). Invasive plants include not only noxious weeds, but also other plants that are not native to the region. The BLM considers plants invasive if they have been introduced into an environment where they did not evolve. Invasive plants usually have no natural enemies to limit their reproduction and spread (Westbrooks 1998). They invade recreation areas, BLM-managed public lands, National Parks, State Parks, roadsides, streambanks, federal, state, and private lands. Invasive weeds can:

- destroy wildlife habitat, reduce opportunities for hunting, fishing, camping and other recreational activities;
- displace RTE species and other species critical to ecosystem functioning (e..g, riparian plants);
- reduce plant and animal diversity;
- invade following wildland and prescribed fire (potentially into previously unaffected areas), limiting regeneration and establishment of native species and rapidly increasing acreage of infested land;
- increase fuel loads and decrease the length of fire cycles and/or increase the intensity of fires;
- disrupt waterfowl and neo-tropical migratory bird flight patterns and nesting habitats; and
- cost millions of dollars in treatment and loss of productivity to private land owners.

The BLM uses an Integrated Pest Management (IPM) approach to manage invasive plants. Management techniques may be biological, mechanical, chemical, or cultural. Many herbicides are currently used by the BLM under their chemical control program. This report considers the impact to ecological receptors (animals and plants) from the use of the herbicide fluridone for the management of aquatic vegetation on BLM lands.

2.2 Herbicide Description

The herbicide-specific use-criteria discussed in this document were obtained from the product label as registered with the USEPA as it applies to the BLM use. Fluridone application rates and methods discussed in this section are based on past and predicted BLM herbicide use and are in accordance with product labels approved by the USEPA. The BLM should be aware of all state-specific label requirements and restrictions. In addition, new USEPA approved herbicide labels may be issued after publication of this report, and BLM land managers should be aware of all newly approved federal, state, and local restrictions on herbicide use when planning vegetation management programs.

Fluridone is a selective systemic herbicide that inhibits carotene production in leaves, which causes the breakdown of chlorophyll—preventing the plant from synthesizing food. This herbicide comes in two formulations: liquid and granule.

Fluridone is being proposed for use in the BLM's Aquatic Vegetation Management program. The majority of application occurs in inland freshwater habitats; diquat is rarely used in marine or estuarine habitats. Applications will be carried out through both aerial and ground application methods. Aerial applications will be made using a fixed-wing airplane or a helicopter. Ground applications will be made on foot, horseback, boat, or using an ATV or truck mounted sprayer applying as a spot or broadcast application. Boat applications will use either a handgun, which will be used to make spot treatments, or a boom, which will be used to make broadcast applications onto the surface of the water or to inject the herbicide under the water surface. The BLM is proposing a typical application rate of 1.0 lbs (lbs) a.i./ac, and the maximum application rate will be 1.3 lbs a.i./ac. Details regarding expected fluridone usage by BLM are provided in Table 2-1 at the end of this section.

2.3 Herbicide Incident Reports

An "ecological incident" occurs when non-target flora or fauna is killed or damaged due to application of a pesticide. When ecological incidents are reported to a state agency or other proper authority, they are investigated and an ecological incident report is generated. The Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) requires product registrants to report adverse effects of their product to the USEPA.

The USEPA OPP manages a database, the EIIS, which contains much of the information in the ecological incident reports. As part of this risk assessment, USEPA was requested to provide all available incident reports in the EIIS that listed fluridone as a potential source of the observed ecological damage.

The USEPA EIIS contained one incident report involving fluridone. Fluridone was listed as the "probable" cause of damage to tomato (*Lycopersicon esculentum*) plants due to direct contact. The type of herbicide use (e.g., registered use, accidental, misuse) and severity of the impact was not specified. There were no other pesticides implicated in this incident report.

TABLE 2-1
BLM Fluridone Use Statistics

Program	Scenario	Vehicle	Method	Used?	Application Rate Typical (lbs a.i./ac)	Application Rate Maximum (lbs a.i./ac)
Rangeland				No		
Public-Domain Forest Land				No		
Energy & Mineral Sites				No		
Rights-of-way				No		
Recreation				No		
Aquatic	Aerial	Plane	Fixed Wing	Yes	0.15	1.3
		Helicopter	Rotary	Yes	0.15	1.3
	Ground	Human	Backpack	Yes	0.15	1.3
			Horseback	Yes	0.15	1.3
		ATV	Spot	Yes	0.15	1.3
			Boom/Broadcast	Yes	0.15	1.3
		Truck	Spot	Yes	0.15	1.3
			Boom/Broadcast	Yes	0.15	1.3

The BLM applies fluridone at different typical and maximum rates for four different water bodies: Ponds, Whole Lake/Reservoir, Partial Lakes/Reservoir, and Canals. The lowest typical application rate (Whole Lake/Reservoir) was selected for use as the typical rate and the highest maximum application rate (Partial Lake/Reservoir) was selected for use as the maximum application rate. Application rates are dependent on water depth, which is assumed to be 1 meter (3.28 feet).

3.0 HERBICIDE TOXICOLOGY, PHYSICAL-CHEMICAL PROPERTIES, AND ENVIRONMENTAL FATE

This section summarizes available herbicide toxicology information, describes how this information was obtained, and provides a basis for the LOC values selected for this risk assessment. Fluridone's physical-chemical properties and environmental fate are also discussed.

3.1 Herbicide Toxicology

A review of the available ecotoxicological literature was conducted in order to evaluate the potential for fluridone to negatively effect the environment and to derive TRVs for use in the ERA (provided in italics in sections 3.1.2 and 3.1.3). The process for the literature review and the TRV derivation is provided in the Methods Document (ENSR 2004c). This review generally included a review of published manuscripts and registration documents, information obtained through a Freedom of Information Act (FOIA) request to EPA, electronic databases (e.g., EPA pesticide ecotoxicology database, EPA's on-line ECOTOX database), and other internet sources. This review included both freshwater and marine/estuarine data, although the focus of the review was on the freshwater habitats more likely to occur on BLM lands.

Endpoints for aquatic receptors and terrestrial plants were reported based on exposure concentrations (mg/L and lbs/ac, respectively). Dose-based endpoints (e.g., LD_{50}s) were used for birds and mammals. When possible, dose-based endpoints were obtained directly from the literature. When dosages were not reported, dietary concentration data were converted to dose-based values (e.g., LC_{50} to LD_{50}) following the methodology recommended in USEPA risk assessment guidelines (Sample et al. 1996). Acute TRVs were derived first to provide an upper boundary for the remaining TRVs; chronic TRVs were always equivalent to, or less than (<), the acute TRV. The chronic TRV was established as the highest NOAEL value that was less than both the chronic lowest observed adverse effect level (LOAEL) and the acute TRV. When acute or chronic toxicity data was unavailable, TRVs were extrapolated from other relevant data using an uncertainty factor of 3, as described in the Methods Document (ENSR 2004c).

This section reviews the available information identified for fluridone and presents the TRVs selected for this risk assessment (Table 3-1). Appendix A presents a summary of the fluridone data identified during the literature review. Toxicity data are presented in the units used in the reviewed study. In most cases this applies to the a.i. itself (e.g., fluridone); however, some data correspond to a specific product or applied mixture (e.g., Sonar) containing the a.i. under consideration, and potentially other ingredients (e.g., other a.i. or inert ingredients). This topic, and others related to the availability of toxicity data, is discussed in Section 7.1 of the Uncertainty section. The review of the toxicity data did not focus on the potential toxic effects of inert ingredients (inerts), adjuvants, surfactants, and degradates. Section 7.3 of the Uncertainty section discusses the potential impacts of these constituents in a qualitative manner.

3.1.1 Overview

According to USEPA ecotoxicity classifications presented in registration materials,[1] fluridone has low toxicity to most terrestrial species. Studies conducted with mammals found that acute exposure to fluridone commonly does not cause adverse effects, even to mammals that were exposed to fluridone for longer periods of time or during

[1] Available at http://www.epa.gov/oppefed1/ecorisk_ders/toera_analysis_eco.htm#Ecotox

pregnancy. Similarly, short-term exposure to fluridone did not result in adverse effects in birds, even at high exposure levels. Long-term exposure to fluridone did result in reduced growth in large and small birds. Fluridone was classified as practically non-toxic to honeybees. While no quantitative data were found to evaluate fluridone's effects on terrestrial plants, the manufacturer's user guide (Eli Lilly and Company 2003) provided qualitative results indicating that the sensitivity of terrestrial plants is variable. Some species (e.g., grasses and sedges) were more sensitive than other plant species (e.g., willow).

Fluridone is an herbicide used to control aquatic plants. In the available literature, aquatic plants were not affected by concentrations up to 1 mg/liter (L) (Anderson 1991). Acute and chronic toxicity tests indicate that fluridone causes toxicity to fish species at concentrations < 10 mg/L, and some adverse effect concentrations approach 1 mg/L (Hamelink et al. 1986). No data were found to evaluate the toxicity of fluridone to amphibians. Acute toxicity concentrations for aquatic invertebrates were as low as 1.3 mg /L (Hamelink et al. 1986), which is equal to the maximum application rate.

3.1.2 Toxicity to Terrestrial Organisms

3.1.2.1 Mammals

Oral toxicity studies conducted in small mammals demonstrated that acute exposure to fluridone typically does not cause adverse effects, even at relatively high dose levels (greater than [>] 10,000 mg a.i./kilogram (kg) body weight (BW) (USEPA 1979). Similarly, acute dermal exposure studies found no adverse effects to rabbits (*Leporidae* spp.) exposed to 5,000 mg a.i./kg BW of fluridone (Eli Lilly 2003). Adverse effects were demonstrated during studies of longer duration. In subchronic oral gavage studies, rabbits exhibited signs of maternal and fetal toxicity (decreased maternal weight, abortions) when dosed with 300 mg a.i./kg BW-day of fluridone during pregnancy (Integrated Risk Information System [IRIS] 2003, MRID 00103302). In this same study, no adverse effects were noted at 125 mg a.i./kg BW-day.

The effects of dietary exposure to fluridone were evaluated in several long-term feeding trails. Rats (*Rattus* spp.) fed fluridone for two years at dietary concentrations as high as 650 parts per million (ppm; equivalent to 25 mg a.i./kg BW-day) exhibited adverse effects, such as decreased BWs and damage to kidneys, testes, and eyes. In this same study, no adverse effects were observed at concentrations of 200 ppm (equivalent to 8 mg a.i./kg BW-day) (IRIS 2003, MRID 00135208).

Based on these findings, the oral LD_{50} (the dose that causes the mortality of 50 percent of the organisms tested; >10,000 mg a.i./kg BW) and chronic dietary NOAEL (8 mg a.i./kg BW-day) were selected as the dietary small mammal TRVs. The dermal small mammal TRV was established at >5,000 mg a.i./kg BW.

For large mammals, a one-year feeding trial showed systemic effects (weight loss, increased liver weight, and alkaline phosphatase) in beagle dogs (*Canis familiaris*) fed 150 mg a.i./kg BW-day, while no adverse effects were observed in dogs fed 75 mg a.i./kg BW-day (CA EPA 2000).

Since no large mammal LD_{50}s were identified in the available literature, the small mammal LD_{50} (>10,000 mg a.i./kg BW) was used as a surrogate value. The large mammal dietary NOAEL TRV was established at 75 mg a.i./kg BW-day.

Overall, acute exposure to fluridone causes few adverse effects to mammals, but adverse effects can occur if mammals are chronically exposed to fluridone. Small mammals may be slightly more susceptible to fluridone than large mammals.

3.1.2.2 Birds

Information related to avian exposure to fluridone suggests that acute oral exposure to fluridone is practically non-toxic to birds. The LD_{50} value (the dose that causes the mortality of 50 percent of the organisms tested) was > 2,000 mg/kg BW for bobwhite quail (*Colinus virginianus*) orally administered technical grade fluridone at 95 to 97% a.i.

(USEPA 2003b). In dietary studies, the LC_{50} for bobwhite quail was reported to be > 4,350 ppm of fluridone (equivalent to a dose of 2,627 mg a.i./kg BW-day) (USEPA 1978). For mallards (*Anas platyrhynchos*), the dietary LC_{50} value for fluridone was > 4,540 ppm (equivalent to 454 mg a.i./kg BW-day) for acute exposures (USEPA 1978). In these dietary tests, the test organism was presented with the dosed food for 5 days, with 3 days of additional observations after the dosed food was removed. The endpoint reported for this assay is generally an LC_{50} representing mg a.i./ kg food. For this ERA, the concentration based value was converted to a dose-based value following the methodology presented in the Methods Document (ENSR 2004c). Then the dose-based value was multiplied by the number of days of exposure (generally 5) to result in an LD_{50} value representing the full herbicide exposure over the course of the test. This resulted in LD_{50} values of >13,135 mg a.i./kg BW and >2,270 mg a.i./kg BW for the bobwhite quail and mallard, respectively. Although this study did not provide information regarding % a.i., it was conducted with technical grade fluridone which is generally 95 to 97% a.i.

Similarly, birds fed high concentrations of fluridone in their diets for longer periods of time also showed no adverse effects. Bobwhite quail exposed to 1,000 ppm of fluridone (equivalent to 604 mg a.i./kg BW-day) via the diet for an entire generation did not exhibit signs of systemic or reproductive adverse effects (USEPA 2003b, ACC070932). Similarly, mallards fed 1,000 ppm fluridone (equivalent to 100 mg a.i./kg BW-day) in their diets for an entire generation did not show signs of adverse effects (USEPA 2003b, ACC070932).

Based on these findings, the bobwhite quail dietary LD_{50} (>13,135 mg/kg BW) and chronic NOAEL (604 mg a.i./kg BW-day) were selected as the small bird dietary TRVs. The mallard dietary LD_{50} (>2,270 mg/kg BW) and NOAEL (100 mg a.i./kg BW-day) were selected as the large bird dietary TRVs.

3.1.2.3 Terrestrial Invertebrates

A standard acute contact toxicity bioassay in honeybees is required for the USEPA pesticide registration process. In this study, fluridone was directly applied to the bee's thorax and mortality was assessed during a 48-hr period. The USEPA reports a NOAEL of 362.58 micrograms (μg)/bee using a 33.3% a.i. technical fluridone product (USEPA 2003b, ACC070932).

In a manufacturer's user's guide (Eli Lilly and Company 2003), data were presented indicating that no mortality has been observed in toxicity tests with earthworms exposed to concentrations as high as 102.6 ppm. This value could not be confirmed by any other source of information reviewed for this document.

Since an LD_{50} was not established in the literature, the NOAEL was multiplied by an uncertainty factor of 3, resulting in a LD_{50} of 1,088 μg/bee. Based on a honeybee weight of 0.093 g, this TRV was expressed as 11,699 mg a.i./kg BW. This uncertainty factor was selected based on a review of the application of uncertainty factors (Chapman et al. 1998), and the use of uncertainty factors for this assessment is described in the Methods Document (ENSR 2004c).

3.1.2.4 Terrestrial Plants

Fluridone is sold commercially as Sonar and is primarily used to control aquatic weeds. No quantitative toxicity studies were found in the reviewed literature that addressed toxicity of fluridone to terrestrial plants. In the manufacturer's user's guide (Eli Lilly and Company 2003), grasses and some sedges are considered to be "sensitive" or "intermediate" in their tolerance to the Sonar herbicide, while rushes tend to be "intermediate" to "tolerant". Shoreline plants, such as willow (*Salix* spp.) and cypress (*Cupressus* spp.), were considered "tolerant," while the tolerance of members of the evening primrose (*Oenothera* and *Camissonia* spp.) and acanthus families (Acanthaceae) was classified as "intermediate".

3.1.3 Toxicity to Aquatic Organisms

3.1.3.1 Fish

In acute toxicity tests, the 96-hour LC_{50} value (i.e., concentration that cause 50% mortality) for rainbow trout (*Oncorhynchus mykiss*) was found to be as low as 4.2 mg/L (Hamelink et al. 1986). Acute toxicity tests conducted on warmwater fish species (bluegill sunfish [*Lepomis macrochirus*], fathead minnow [*Pimephales promelas*], and channel catfish [*Ictalurus punctatus*]) documented 96-hour LC_{50} values as low as 8.2 mg/L (Hamelink et al. 1986; USEPA 2003b, MRID 40098001). Chronic, life-cycle tests on fathead minnow showed adverse effects at fluridone concentrations of 0.96 mg/L, and no adverse effects at concentrations of 0.48 mg /L (Hamelink et al. 1986, USEPA, 2003b, ACC 070934). As a consequence, fluridone is considered to be moderately toxic to fish species. Most studies reviewed, and all studies selected, for TRV derivation for fish were based on products containing at least 97% fluridone.

The lower of the cold- and warmwater fish endpoints were selected as the TRVs for fish. Therefore the coldwater 96-hour LC_{50} of 4.2 mg a.i./L was selected as the acute TRV, and the warmwater fish NOAEL of 0.48 mg a.i./L was used as the TRV for chronic effects.

3.1.3.2 Amphibians

No toxicity studies for amphibians were found in the literature reviewed for this document.

3.1.3.3 Aquatic Invertebrates

The toxicity of fluridone was evaluated with several freshwater aquatic invertebrates, including water fleas (e.g., *Daphnia magna*), scuds (*Hyallela* spp.), crayfish (e.g. Astacidae), and chironomids. Acute toxicity was observed in aquatic invertebrates exposure to fluridone concentrations as low as 1.3 mg/L (Hamelink et al. 1986; USEPA 2003b, MRID 40098001). This result is listed for several different studies with % a.i. ranging from 41% to 98% fluridone. Based on the available information, crayfish appear to be less sensitive than other aquatic invertebrates, with LC_{50}s above 16.9 mg a.i./L (Hamelink et al. 1986). NOAELs for several species were derived from chronic or short-term chronic studies. The 21 day reproduction NOAEL for *D. magna* is 0.2 mg/L and the chronic NOAELs for *Gammarus pseudolimnaeus* (60 day growth endpoint) and *Chironomus plumosus* (30 day emergence endpoint) is 0.6 mg/L using a technical grade fluridone at 98 to 99% a.i. (Hamelink et al. 1986).

The LC_{50} (1.3 mg/L) was selected as the invertebrate acute TRV, and the NOAEL of 0.6 mg/L was selected as the chronic TRV.

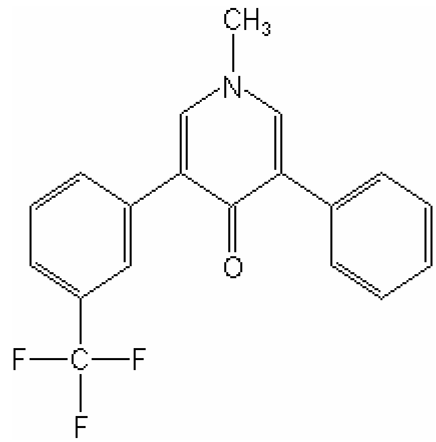
3.1.3.4 Aquatic Plants

Standard toxicity tests were conducted on aquatic plants. The duration of the studies ranged from 37 days to 15 months (McCowen et al. 1979; Anderson 1981; Farone & McNabb 1993; Netherland et al. 1997; Madsen et al. 2002). Study endpoints evaluated included species diversity and growth, measured as biomass and length. Studies failed to detect adverse effects to aquatic macrophytes with fluridone concentrations as high as 1 mg/L (Anderson. 1991). No information was provided regarding the % fluridone contained in the tested product, although it is identified as fluridone [1-methyl-3-phenyl-5-[3-(trifluoromethyl)phenyl]-4(1H)-pyridinone applied at 9.3 liters (L) per hectare (Anderson. 1991).

The NOAEL was set at 1 mg/L. Since no Median Effective Concentration [EC_{50}] values were identified in the reviewed literature, the NOAEL was multiplied by an uncertainty factor of 3 to estimate an EC_{50} of 3 mg./L.

3.2 Herbicide Physical-Chemical Properties

The chemical formula for fluridone is 1-methyl-3-phenyl-5-(α,α,α-trifluoro-m-tolyl)-4-pyridone. At low pH values, some of the fluridone molecules will exist as cations (pKa = 1.7) (Reinert 1989). The chemical structure of fluridone is shown below:



Fluridone Chemical Structure

The physical-chemical properties and degradation rates critical to fluridone's environmental fate are listed in Table 3-2, which presents the range of values encountered in the literature for these parameters. To complete Table 3-2, available USEPA literature on fluridone was obtained either from the Internet or through a FOIA request. Herbicide information that had not been cleared of Confidential Business Information (CBI) was not provided by USEPA as part of the FOIA documents. Additional sources, both on-line and in print, were consulted for information about the herbicide:

- The British Crop Protection Council and The Royal Society of Chemistry. 1994. The Pesticide Manual Incorporating the Agrochemicals Handbook. Tenth Edition. Surrey and Cambridge, United Kingdom.

- California Department of Pesticide Registration (DPR.). 2003. USEPA/OPP Pesticide Related Database. Updated weekly. Available at: http://www.cdpr.ca.gov/docs/epa/epamenu.htm.

- Compendium of Pesticide Common Names. 2003. A website listing all International Organization for Standardization (ISO)-approved names of chemical pesticides. Available at: http://www.hclrss.demon.co.uk.

- Hazardous Substances Data Bank (HSDB). 2002. A toxicology data file on the National Library of Medicines Toxicology Data Network (TOXNET). Available at: http://toxnet.nlm.nih.gov.

- Hornsby, A., R. Wauchope, and A. Herner. 1996. Pesticide Properties in the Environment. P. Howard (ed.). Springer-Verlag, New York.

- Mackay, D., S. Wan-Ying, and M. Kuo-ching. 1997. Handbook of Environmental Fate and Exposure Data for Organic Chemicals. Volume III. Pesticides Lewis Publishers, Chelsea, Minnesota.

- Montgomery, J.H. (ed.). 1997. Illustrated Handbook of Physical-Chemical Properties and Environmental Fate for Organic Chemicals. Volume V. Pesticide Chemicals. Lewis Publishers, Boca Raton, Florida.

- Tomlin, C (ed.). 1994. The Agrochemicals Desk Reference 2nd Edition. Lewis Publishers, Boca Raton, Florida.

In addition, information was also obtained from the product label for the herbicide Sonar A.S. (SePRO 2002a), the Handbook of Environmental Degradation Rates (Howard et al. 1991), and a fact sheet prepared by Washington State's Department of Health (WA Dept of Health 2000). Relevant papers from the scientific literature were also reviewed. These papers were obtained as part of the literature review to define ecological toxicity endpoints. Values for the foliar half-life and for the foliar washoff coefficient were not found during the review of chemical-physical properties. Thus, as conservative estimates, a foliar half-life of 365 days (no herbicide degradation occurs while on foliage) and a foliar washoff fraction of 1 (all herbicide washes off plant during the first rain) were used in risk assessment calculations. The half-life in pond water was estimated using the physical-chemical properties listed in Table 3-2 and the information reviewed concerning the environmental fate of fluridone in aquatic systems. Values for foliar half-life and foliar washoff fraction were obtained from a database included in the Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) computer model (United States Department of Agriculture [USDA] 1999). Residue rates were obtained from the Kenaga nomogram, as updated (Fletcher et al. 1994). Values selected for use in risk assessment calculations are shown in bold in Table 3-2, presented at the end of this section.

3.3 Herbicide Environmental Fate

The Pesticide Manual reports that biodegradation is the primary fluridone loss mechanism from soils (The British Crop Protection Council and The Royal Society of Chemistry 1994). Soil biodegradation half-lives from 44 days to 192 days have been reported (Howard et al. 1991). The K_{oc}, or organic carbon-water partitioning coefficient, measures the affinity of a chemical to organic carbon relative to water. The higher the K_{oc}, the less soluble in water and the higher affinity for organic carbon, an important constituent of soil particles. Therefore, the higher the K_{oc}, the less mobile the chemical. All but one of the K_{oc} values reviewed ranged from 270 to 6400, indicating fluridone has moderate to no mobility in soils (Table 3-2; Swann et al. 1986). Fluridone sorption increases with clay content, organic matter content, cation exchange capacity, surface area, and decreasing pH (Table 3-2; Weber et al. 1986; Reinert 1989). Protonation at low pH values leads to increased sorption due to cation exchange (Reinert 1989). Fluridone is stable to hydrolysis (USEPA 1986). Based on its Henry's Law constant (the ratio of the chemical's equilibrium distribution between the gas and liquid phases) and vapor pressure, fluridone might volatilize slowly from wet soil surfaces, but volatilization from dry soils would not be expected (Lyman et al. 1990; Mackay et al. 1997; HSDB 2002; Table 3-2). Field half-lives ranging from 21 days to five years have been reported (Table 3-2).

In aquatic systems, photodegradation and biodegradation are important loss pathways for fluridone (The British Crop Protection Council and The Royal Society of Chemistry 1994). As in terrestrial systems, fluridone is stable to hydrolysis and based on the Henry's law constant would volatilize slowly from water bodies (USEPA 1986; Lyman et al. 1990; Mackay et al. 1997; HSDB 2002; Table 3-2). Also, based on reported K_{oc} values, fluridone would be expected to sorb to suspended solids and sediments in aquatic systems (Tomlin 1994). Desorption from sediments followed by photolysis is reported to be a major loss mechanism from aquatic systems (Tomlin 1994). Biodegradation may also remove fluridone from aquatic systems (WA Department of Health 2000). Based on a bioconcentration factor (BCF) of 3.01, fluridone would have little tendency to bioaccumulate in fish (Table 3-2; WA Department of Health 2000). Aquatic dissipation half-lives from 4 to 7 days to 9 months (anaerobic sediments) have been reported (Table 3-2).

TABLE 3-1
Selected Toxicity Reference Values for Fluridone

Receptor	Selected TRV	Units	Duration	Endpoint	Species	Notes
RECEPTORS INCLUDED IN FOOD WEB MODEL						
Terrestrial Animals						
Honeybee	1,088	µg/bee	48 h	LD_{50}		extrapolated from NOAEL; 33.3% a.i. product
Large bird	> 2,270	mg/kg bw	8 d	LD_{50}	mallard	technical grade; assumed 95 - 97% a.i.
Large bird	100	mg a.i./kg bw-day	1 generation	NOAEL	mallard	reproduction
Piscivorous bird	100	mg a.i./kg bw-day	1 generation	NOAEL	mallard	
Small bird	> 13,135	mg/kg bw	8 d	LD_{50}	bobwhite quail	technical grade; assumed 95 - 97% a.i.
Small bird	604	mg a.i./kg bw-day	1 generation	NOAEL	bobwhite quail	reproduction
Small mammal	8	mg a.i./kg bw-day	2 y	NOAEL	rat	
Small mammal - dermal	> 5,000	mg a.i./kg bw	8 d	LD_{50}	rabbit	
Small mammal - ingestion	> 10,000	mg a.i./kg bw	NR	LD_{50}	mouse and rat	water exposure; no diet available
Large mammal	> 10,000	mg a.i./kg bw	NR	LD_{50}	mouse and rat	small mammal value
Large mammal	75	mg a.i./kg bw-day	1 y	NOAEL	beagle	
Terrestrial Plants						
Terrestrial plants -typical species	no data					
Terrestrial plants - RTE species	no data					
Aquatic Species						
Aquatic invertebrates	1.3	mg/L	48 h	LC_{50}	midge (*Chironomus*)	multiple studies; 41% - 98% a.i.
Fish	4.25	mg/L	96 h	LC_{50}	rainbow trout	98 – 99% a.i. product
Aquatic plants and algae	3	mg/L	37 d	EC_{50}	American pondweed	extrapolated from NOAEL; no % a.i. listed
Aquatic invertebrates	0.6	mg/L	30 d	NOAEL	midge (*Chironomus*)	98 – 99% a.i. product
Fish	0.48	mg/L	life cycle	NOAEL	fathead minnow	extrapolated from LOAEL; swimming speed
Aquatic plants and algae	1	mg/L	37 d	NOAEL	American pondweed	biomass

TABLE 3-1 (Cont.)
Selected Toxicity Reference Values for Fluridone

Receptor	Selected TRV	Units	Duration	Endpoint	Species	Notes
ADDITIONAL ENDPOINTS						
Amphibian	no data					
Amphibian	no data					
Warmwater fish	8.2	mg/L	96 h	LC_{50}	channel catfish	98 – 99% a.i. product
Warmwater fish	0.5	mg/L	life cycle	NOAEL	fathead minnow	98 – 99% a.i. product
Coldwater fish	4.2	mg/L	96 h	LC_{50}	rainbow trout	98 – 99% a.i. product
Coldwater fish	1.4	mg/L	96 h	NOAEL	rainbow trout	extrapolated from LC_{50}

Notes:
Toxicity endpoints for terrestrial animals
LD_{50} - to address acute exposure.
NOAEL - to address chronic exposure.
Toxicity endpoints for terrestrial plants
EC_{25} - to address direct spray, drift, and dust impacts on typical species.
EC_{05} or NOAEL - to address direct spray, drift, and dust impacts on threatened or endangered species.
Toxicity endpoints for aquatic receptors
LC_{50} or EC_{50} - to address acute exposure (appropriate toxicity endpoint for non-target aquatic plants will be an EC50).
NOAEL - to address chronic exposure.
Value for fish is the lower of the warmwater and coldwater values.

Piscivorous bird TRV = Large bird chronic TRV.
Fish TRV = lower of coldwater and warm water fish TRVs.
Durations:
h - hours
d - days
w - weeks
m - months
y - years
NR – Not reported
Units represent those presented in the reviewed study

TABLE 3-2
Physical-Chemical Properties of Fluridone

Parameter	Value
Herbicide family	Unclassified herbicide (Compendium of Pesticide Common Names 2003).
Mode of action	Inhibits carotene production, which leads to chlorophyll breakdown. (SePRO 2002a).
Chemical Abstract Service number	59756-60-4 (Mackay et al. 1997).
Office of Pesticide Programs chemical code	112900 (DPR 2003).
Chemical name (International Union of Pure and Applied Chemistry [IUPAC])	1-methyl-3-phenyl-5-(α,α,α-trifluoro-m-tolyl)-4-pyridone (Tomlin 1994).
Empirical formula	$C_{19}H_{14}F_3NO$ (Mackay et al. 1997).
Molecular weight (MW)	**329.3** (Tomlin 1994).
Appearance, ambient conditions	White to tan crystalline solid (technical product) (Tomlin 1994).
Acid / Base properties	1.7 (pKa) (Reinert 1989).
Vapor pressure (millimeters of mercury [mmHg] at 25ºC)	< $1x10^{-7}$ (Weber et al. 1986); 9.8×10^{-8} (Mackay et al. 1997; Tomlin 1994); 1×10^{-7} (Hornsby 1996).
Water solubility (mg/L at 25ºC)	12 (Reinert 1989); 12 (pH 7) (Mackay et al. 1997; Tomlin 1994); 10 (Hornsby et al. 1996).
Log Octanol-water partition coefficient (Log(K_{OW}), unitless)	**1.87** (pH 7, 25ºC) (Tomlin 1994; USEPA 1982); 2.98 (Mackay et al. 1997).
Henry's law constant (atm-m^3/mole)	3.52×10^{-6} (Mackay et al. 1997).
Soil / Organic matter sorption coefficient (Kd / K_{oc})	880 (K_{oc}). K_{oc} values from 70 to 2700 obtained for three soils. Kd (Freundlich) / K_{oc} for three soils: 29 / 2700 (Stockton clay, pH 6, organic matter 1.8%, clay 60%, cation exchange capacity 44), 8.6 / 370 (Yolo sandy clay loam, pH 7, organic matter 4.0%, clay 21%, cation exchange capacity 21), and 2.7 / 270 (Hesperia fine sandy loam, pH 7.3, organic matter 1.7%, clay 8.5%, cation exchange capacity 8.5) (Reinert 1989). Freundlich Kd values of 2.6-38 measured on 13 soils (Weber et al. 1986). All values are log(K_{oc}): 2.544-3.04, 1.60 (soil), 2.97-3.39 (pond sediment), 3.36, 2.95 (lake and river sediment), 3.00 (Mackay et al. 1997). For 5 soils, 3-16 (Kd), 350-1100 (K_{oc}) (Tomlin 1994); 1000 (K_{oc}) (Hornsby et al. 1996).
Bioconcentration factor (BCF)	175 samples, 10 fish species: Whole fish BCF for fluridone = **3.01** (West et al. 1983; USEPA 1982).
Field dissipation half-life	6 months to 5 years (USEPA 1982); Ranging from 46-365 days observed for fluridone applied at 1 or 10 μg ai/g soil on sandy loam, sandy clay loam, and peaty loam soils at three different moisture contents (1/4 field capacity, 1/2 field capacity, field capacity, and wet-dry cycling) and two temperature regimes (10ºC and 18-24ºC). Longest half-life generally found for driest condition (Malik 1990); 21 days (Hornsby et al. 1996).
Soil dissipation half-life(1)	Estimated 103 and 27 days (based on dissipation rates of 0.0067 and 0.025 1/day) (Mackay et al. 1997); In a silt-loam > 343 days (pH 7.3, organic matter 2.6%) (Tomlin 1994); Soil aerobic of 44-192 days based on soil die-away test data and field study soil persistence (Howard et al. 1991).
Aquatic dissipation half-life	Fluridone concentration decreased logarithmically with time after Sonar 4AS treatment (liquid) in two NC ponds at 1.0 lb ai/ac and 2.0 lbs ai/ac. Estimated time to reach zero concentration, 64 and 69 days. No observed decrease in a VA pond treated with Sonar 5P, a pelleted formulation, for 53 days (1.0 lb ai/ac). Authors speculate that shading in pond receiving Sonar 5P reduced loss due to photolysis (Langeland and Warner 1986). Half-lives ranging from 4-7 days reported for fluridone in Canadian fish ponds applied at 70, 700, and 5000 μg ai/L (Muir et al. 1980). 5-60 days (av. 20) in 13 ponds treated with SONAR AS: Pond locations FL, TX, TN, CA, WV, IN, MO, MI, NY, and Manotick, Canada. Ponds treated with SONAR 5P (pelleted) reached max fluridone concentration ~ 14 days.

TABLE 3-3 (Cont.)
Physical-Chemical Properties of Fluridone

Parameter	Value
Aquatic dissipation half-life (continued)	after treatment and then fluridone levels declined at a rate similar to ponds treated with SONAR AS. Lake half-lives less than 1 week due to dispersion and dilution as well as degradation and/or adsorption (West et al. 1983); Hydrosoil degradation product only observed in laboratory experiments. In aquatic systems, no degradate observed. Believed desorption followed by photolysis responsible for loss from sediments. In ponds treated with SONAR AS, hydrosoil concentrations reached max after ~ 1 month. In SONAR AP treated ponds, hydrosoil concentrations reached a max within 14 days after treatment. Average half-life for declining phase of fluridone in hydrosoils of SONAR AS treated ponds was 3 months. No fluridone found in treated lake sediments. (West et al. 1983); 21 days in surface water (Mackay et al. 1997); In water (anaerobic) 9 months, (aerobic) about 20 days. (Tomlin 1994; USEPA 1982); Surface water 12-36 days based upon estimated photolysis in water, ground water 88-383 days based upon estimated unacclimated aqueous aerobic biodegradation (Howard et al. 1991).
Hydrolysis half-life	Stable to hydrolysis (USEPA 1986); Stable to hydrolysis, pH = 3 to 9. (Tomlin 1994); > 113 days for 1 μg/ml to hydrolyze in pond water at 4ºC (Mackay et al. 1997).
Photodegradation half-life in water	26 - 55 hours (pH 3 to 9, different fluridone concentrations, pond water, distilled water, no oxygen water) (USEPA 1982); ~ 23 hours in distilled water under > 290 nm light, ~6 hours for 5 ug/ml to degrade in nonsterile pond water under sunlight, ~27 days for 85% of 10 μg/ml to degrade in distilled water and for 85% of 10 ug/ml to degrade in lake water at pH 8.4 both under sunlight (Mackay et al. 1997); 12-36 days based upon measured rate constant for summer sunlight photolysis in distilled water (12 days) and adjusted for relative winter sunlight intensity (36 days) (Howard et al. 1991).
Photodegradation half-life in soil	Not available.
Soil biodegradation half-life	Soil aerobic of 44-192 days based on soil die-away test data and field study soil persistence (Howard et al. 1991).
Aquatic biodegradation half-life	In aquatic systems: 20 days (aerobic), 9 months (anaerobic), 90 days (hydrosoil) (USEPA 1986).
Other degradation rates / half-lives	In hydrosoil > 1 year after initial application and 20 weeks in a retreated pond (Muir et al. 1980).
Foliar half-life	not available.[2]
Residue Rate for grass [3]	**197 ppm** (maximum) and **36 ppm** (typical) per lb a.i./ac
Residue Rate for vegetation [4]	**296 ppm** (maximum) and **35 ppm** (typical)
Residue Rate for insects [5]	**350 ppm** (maximum) and **45 ppm** (typical)
Residue Rate for berries [6]	**40.7 ppm** (maximum) and **5.4 ppm** (typical)

Notes:
Values presented in bold were used in risk assessment calculations.
(1) Some studies listed in this category may have been performed under field conditions, but insufficient information was provided in the source material to make this determination.
(2) A foliar half-life was not found during our literature review and the available information concerning fluridone's environmental fate did not suggest a value that could be used as a reasonable surrogate. As a conservative estimate, the foliar half-life of fluridone was set at **365 days** for use in risk assessment calculations; that is, fluridone degradation is zero on the time scale of the simulation.
(3) Residue rates selected are the high and mean values for long grass. Fletcher et al. (1994).
(4) Residue rates selected are the high and mean values for leaves and leafy crops. Fletcher et al. (1994).
(5) Residue rates selected are the high and mean values for forage such as legumes. Fletcher et al. (1994).
(6) Residue rates selected are the high and mean values for fruit (includes both woody and herbaceous). Fletcher et al. (1994).

4.0 ECOLOGICAL RISK ASSESSMENT

This section presents a screening-level evaluation of the risks to ecological receptors from potential exposure to the herbicide fluridone. The general approach and analytical methods for conducting the fluridone ERA were based on the USEPA's Guidelines for ERA (hereafter referred to as the "Guidelines;" USEPA 1998).

The ERA is a structured evaluation of all currently available scientific data (exposure chemistry, fate and transport, toxicity, etc.) that leads to quantitative estimates of risk from environmental stressors to non-human organisms and ecosystems. The current Guidelines for conducting ERAs include three primary phases: problem formulation, analysis, and risk characterization. These phases are discussed in detail in the Methods Document (ENSR 2004c) and briefly in the following sub-sections.

4.1 Problem Formulation

Problem formulation is the initial step of the standard ERA process and provides the basis for decisions regarding the scope and objectives of the evaluation. The problem formulation phase for fluridone assessment included:

- definition of risk assessment objectives;
- ecological characterization;
- exposure pathway evaluation;
- definition of data evaluated in the ERA;
- identification of risk characterization endpoints; and
- development of the conceptual model.

4.1.1 Definition of Risk Assessment Objectives

The primary objective of this ERA was to evaluate the potential ecological risks from fluridone to the health and welfare of plants and animals and their habitats. This analysis is part of the process used by the BLM to determine which of the proposed treatment alternatives evaluated in the EIS should be used on BLM-managed lands.

An additional goal of this process was to provide risk managers with a tool that develops a range of generic risk estimates that vary as a function of site conditions. This tool primarily consists of Excel spreadsheets (presented in the Ecological Risk Assessment Worksheets; Appendix B), which may be used to calculate exposure concentrations and evaluate potential risks in the risk assessment. A number of the variables included in the worksheets can be modified by BLM land managers for future evaluations.

4.1.2 Ecological Characterization

As described in Section 2.2, fluridone is used by the BLM for vegetation control in Aquatic program. The proposed BLM program involves the general use and application of herbicides on public lands in 17 western states in the continental US and Alaska. These applications have the potential to affect organisms in a wide variety of ecological habitats that could include: deserts and prairie land, and many others. It is not feasible to characterize all of the potential affected habitats within this report; however, this ERA was designed to address generic receptors, including RTE species (see Section 6.0) that could occur within a variety of habitats.

4.1.3 Exposure Pathway Evaluation

The following ecological receptor groups were evaluated:

- terrestrial animals;
- non-target terrestrial plants; and
- aquatic species (fish, invertebrates, and non-target aquatic plants).

These groups of receptor species were selected for evaluation because they: (1) are potentially exposed to herbicides within BLM management areas (directly or indirectly); (2) are likely to play key roles in site ecosystems; (3) have complex life cycles; (4) represent a range of trophic levels; and (5) are surrogates for other species likely to be found on BLM-managed lands.

The exposure scenarios considered in the ERA were primarily organized by potential exposure pathways. In general, the exposure scenarios describe how a particular receptor group may be exposed to the herbicide as a result of a particular exposure pathway. These exposure scenarios were developed to address potential acute and chronic impacts to receptors under a variety of exposure conditions that may occur within BLM-managed lands. Fluridone is an aquatic herbicide; therefore, as discussed in detail in the Methods Document (ENSR 2004c), the following exposure scenarios were considered:

- direct contact with the herbicide or a contaminated waterbody;
- indirect contact with contaminated foliage;
- ingestion of contaminated food items;
- off-site drift of spray to terrestrial areas; and
- accidental spills to waterbodies.

Two generic waterbodies were considered in this ERA: 1) a small pond (1/4 acre pond of 1 meter [m] depth, resulting in a volume of 1,011,715 L) and 2) a small stream representative of Pacific Northwest low-order streams that provide habitat for critical life-stages of anadromous salmonids. The stream size was established at 2 m wide and 0.2 m deep with a mean water velocity of approximately 0.3 meters per second, resulting in a base flow discharge of 0.12 cubic meters per second (cms).

4.1.4 Definition of Data Evaluated in the ERA

Herbicide concentrations used in the ERA were based on typical and maximum application rates provided by the BLM (Table 2-1). These application rates were used to predict herbicide concentrations in various environmental media (e.g., soils, water). For the aquatic herbicides these calculations were fairly straightforward and generally required only simple algebraic calculations (e.g., water concentrations from direct aerial spray). However, off-site herbicide transport due to spray drift was modeled using the AgDRIFT® computer model. AgDRIFT® Version 2.0.05 (SDTF 2002) is a product of the Cooperative Research and Development Agreement between the USEPA's Office of Research and Development and the Spray Drift Task Force (SDTF, a coalition of pesticide registrants).

4.1.5 Identification of Risk Characterization Endpoints

Assessment endpoints and associated measures of effect were selected to evaluate whether populations of ecological receptors are potentially at risk from exposure to proposed BLM applications of fluridone. The selection process is discussed in detail in Methods Document (ENSR 2004c), and the selected endpoints are presented below (impacts to RTE species are discussed in more detail in Section 6.0).

<u>Assessment Endpoint 1: Acute mortality to mammals, birds, invertebrates, non-target plants</u>

- **Measures of Effect** included median lethal effect concentrations (e.g., LD_{50} and LC_{50}) from acute toxicity tests on target organisms or suitable surrogates. To add conservatism to the RTE assessment, lowest available germination NOAELs were used to evaluate non-target RTE plants, and LOCs for RTE species were lower than for typical species.

Assessment Endpoint 2: Acute mortality to fish, aquatic invertebrates, and aquatic plants

- **Measures of Effect** included median lethal effect concentrations (e.g., LC_{50} and EC_{50}) from acute toxicity tests on target organisms or suitable surrogates (e.g., data from other coldwater fish to represent threatened and endangered salmonids). As with terrestrial species, lowest available germination NOAELs were used to evaluate non-target RTE plants, and LOCs for RTE species were lower than for typical species.

Assessment Endpoint 3: Adverse direct effects on growth, reproduction, or other ecologically important sublethal processes

- **Measures of Effect** included standard chronic toxicity test endpoints such as the no observable adverse effect level (NOAEL) for both terrestrial and aquatic organisms. Depending on data available for a given herbicide, chronic endpoints reflect either individual impacts (e.g., growth, physiological impairment, behavior) or population-level impacts (e.g., reproduction; Barnthouse 1993). For salmonids, careful attention was paid to smoltification (i.e., development of tolerance to seawater and other indications of change of parr [freshwater stage salmonids] to adulthood), thermoregulation (i.e., ability to maintain body temperature), and migratory behavior, if such data were available.

Assessment Endpoint 4: Adverse indirect effects on the survival, growth, or reproduction of salmonid fish

- **Measures of Effect** for this assessment endpoint depended on the availability of appropriate scientific data. Unless literature studies were found that explicitly evaluated the indirect effects of fluridone on salmonids and their habitat, only qualitative estimates of indirect effects were possible. Such qualitative estimates were limited to a general evaluation of the potential risks to food (typically represented by acute and/or chronic toxicity to aquatic invertebrates) and cover (typically represented by potential for destruction of riparian vegetation). Similar approaches are already being applied by USEPA OPP for Endangered Species Effects Determinations and Consultations (http://www.epa.gov/oppfead1/endanger/effects).

4.1.6 Development of the Conceptual Model

The fluridone conceptual model (Figure 4-1) is presented as a series of working hypotheses about how fluridone might pose hazards to the ecosystem and ecological receptors. The conceptual model indicates the possible exposure pathways for the herbicide as well as the types receptors that were evaluated for each exposure pathway. Figure 4-2 presents the trophic levels and receptor groups evaluated in the ERA.

The conceptual model for herbicide application on BLM lands is designed to display potential herbicide exposure through several pathways, although all pathways may not exist for all locations. The exposure pathways and ecological receptor groups considered in the conceptual model are also described in Section 4.1.3.

The aquatic herbicide conceptual model (Figure 4-1) presents essentially three mechanisms for the release of an herbicide into the environment: direct spray (either accidental or during normal applications), drift, and accidental spills. These release mechanisms may occur as the aquatic herbicide is applied to the intended pond area from a boat or from the shoreline. The aquatic herbicide considered in this risk assessment is not applied to streams.

As indicated in the conceptual model figure, accidental direct spray of terrestrial receptors may occur when the aquatic herbicide is being applied from a boat. This may result in herbicide exposure for wildlife or non-target terrestrial plants if they are directly sprayed during the application. Terrestrial wildlife may also be exposed to the herbicide by brushing against sprayed vegetation or by ingesting contaminated food items.

Direct spray of non-target receptors may also occur during shoreline applications of the aquatic herbicide. Herbicides may be applied to either a pond (normal application) or a stream (accidental application) resulting in exposure of aquatic plants, fish, and aquatic invertebrates to impacted water. Piscivorous birds may also be impacted by ingesting contaminated fish from an exposed pond.

During normal application of aquatic herbicides, it is possible for a portion of the herbicide to drift outside of the treatment area and deposit onto non-target terrestrial receptors. This may occur during terrestrial or aerial applications and may result in exposure of non-target terrestrial plants to the aquatic herbicide.

Accidental spills may also occur during normal herbicide applications. Spills represent the worst-case transport mechanism for herbicide exposure. An accidental spill to a waterbody would result in exposure for aquatic plants, fish, and aquatic invertebrates to impacted water.

4.2 Analysis Phase

The analysis phase of an ERA consists of two principal steps: the characterization of exposure and the characterization of ecological effects. The exposure characterization described the source, fate, and distribution of the herbicides in various environmental media. All EECs are presented in Appendix B. The ecological effects characterization consisted of compiling exposure-response relationships from all available toxicity studies on the herbicide.

4.2.1 Characterization of Exposure

The BLM uses herbicides in the Aquatics program with several different application methods (e.g., boat, plane, helicopter). In order to assess the potential ecological impacts of these herbicide uses, a variety of exposure scenarios were considered. These scenarios, which were selected based on actual BLM herbicide usage under a variety of conditions, are described in Section 4.1.3.

When considering the exposure scenarios and the associated predicted concentrations, it is important to recall that the frequency and duration of the various scenarios are not equal. For example, exposures associated with accidental spills will be very rare, while ingestion of contaminated vegetation may be more common. Similarly, direct spray events will be short-lived while ingestion of fish from a contaminated pond may occur over weeks or months following application. The ERA has generally treated these differences in a conservative manner (i.e., potential risks are presented despite their likely rarity and/or transience). Thus, tables and figures summarizing RQs may present both relatively common and very rare exposure scenarios. Additional perspective on the frequency and duration of exposures are provided in the narrative below.

As described in Section 4.1.3, the following ecological receptor groups were selected to address the potential risks due to unintended exposure to fluridone: terrestrial animals, terrestrial plants, and aquatic species. A set of generic terrestrial animal receptors, listed below, were selected to cover a variety of species and feeding guilds that might be found on BLM-managed lands. Unless otherwise noted, receptor BWs were selected from the *Wildlife Exposure Factors Handbook* (USEPA 1993a). This list includes surrogate species, although not all of these surrogate species will be present within each actual application area:

- A pollinating insect with a BW of 0.093 grams (g). The honeybee (*Apis mellifera*) was selected as the surrogate species to represent pollinating insects. This BW was based on the estimated weight of receptors required for testing in 40CFR158.590.

- A small mammal with a BW of 20 g that feeds on fruit (e.g., berries). The deer mouse (*Peromyscus maniculatus*) was selected as the surrogate species to represent small mammalian omnivores consuming berries.

- A large mammal with a BW of 70 kg that feeds on plants. The mule deer (*Odocolieus hemionus*) was selected as the surrogate species to represent large mammalian herbivores, including wild horses and burros (Hurt and Grossenheider 1976).

- A large mammal with a BW of 12 kg that feeds on small mammals. The coyote (*Canis latrans*) was selected as the surrogate species to represent large mammalian carnivores (Hurt and Grossenheider 1976).

- A small bird with a BW of 80 g that feeds on insects. The American robin (*Turdus migratorius*) was selected as the surrogate species to represent small avian insectivores.

- A large bird with a BW of approximately 3.5 kg that feeds on vegetation. The Canada goose (*Branta canadensis*) was selected as the surrogate species to represent large avian herbivores.

- A large bird with a BW of approximately 5 kg that feeds on fish. The Northern subspecies of the bald eagle (*Haliaeetus leucocephalus alascanus*) was selected as the surrogate species to represent large avian piscivores (Brown and Amadon 1968[2]).

Potential impacts to non-target terrestrial plants could not be evaluated quantitatively for fluridone due to a lack of terrestrial plant toxicity data. Aquatic exposure pathways were evaluated using fish, aquatic invertebrates, and non-target aquatic plants in a pond or stream habitat (as defined in Section 4.1.3). Rainbow trout and walleyes (*Stizostedion vitreum*) were surrogates for fish, the water flea and water scud were surrogates for aquatic invertebrates, and non target aquatic plants and algae were represented by giant duckweed (*Spirodela polyrhiza*).

Section 3.0 of the Methods Document (ENSR 2004b) presents the details of the exposure scenarios considered in the risk assessments. The following sub-sections describe the scenarios that were evaluated for fluridone.

4.2.1.1 Direct Spray

Plant and wildlife species may be unintentionally impacted during normal application of an aquatic herbicide as a result of a direct spray of the receptor or the waterbody inhabited by the receptor, indirect contact with dislodgeable foliar residue after herbicide application, or consumption of prey items sprayed during application. These exposures may occur within the application area (direct spray of waterbody) or outside of the application area (consumption of terrestrial prey items accidentally sprayed by aquatic herbicide). Generally, impacts outside of the intended application area are accidental exposures and are not typical of BLM application practices. The following direct spray scenarios were evaluated:

<u>Exposure Scenarios Within the Application Area</u>

- Direct Spray to Pond (normal application)

- Consumption of Fish From Contaminated Pond

<u>Exposure Scenarios Outside the Application Area</u>

- Accidental Direct Spray of Terrestrial Wildlife

- Accidental Direct Spray of Non-Target Terrestrial Plants

- Indirect Contact With Foliage After Accidental Direct Spray

[2] As cited on the Virginia Tech Conservation Management Institute Endangered Species Information System website (http://fwie.fw.vt.edu/WWW/esis/).

- Ingestion of Prey Items Contaminated by Accidental Direct Spray
- Accidental Direct Spray Over Stream (fluridone is not indicated for use in streams)

4.2.1.2 Off-site Drift

During normal application of aquatic herbicides, it is possible for a portion of the herbicide to drift outside of the treatment area and deposit onto non-target terrestrial receptors. To simulate off-site herbicide transport as spray drift, AgDRIFT® software was used to evaluate a number of possible scenarios. Based on actual BLM uses of fluridone, ground applications were modeled using a low- or high-placed boom and aerial application was modeled from both a helicopter and a plane over non-forested land. Ground applications were modeled using either a high boom (spray boom height set at 50 inches above the ground) or a low boom (spray boom height set at 20 inches above the ground). Deposition rates vary by the height of the application (the higher the application height, the greater the off-target drift). Drift deposition was modeled at 25, 100, and 900 feet (ft) from the application area for ground applications and 100, 300, and 900 ft from the application area for aerial applications. The AgDRIFT® model determined the fraction of the application rate that is deposited off-site without considering herbicide degradation. Impacts to off-site terrestrial plants were evaluated based on deposition modeled by AgDRIFT.®

4.2.1.3 Accidental Spill to Pond

To represent worst-case potential impacts to the pond, two spill scenarios were considered. These consist of a truck or a helicopter spilling entire loads (200 gallon [gal] spill and 140 gal spill, respectively) of herbicide mixed for the maximum application rate into the 1/4 acre, 1 meter deep pond.

4.2.2 Effects Characterization

The ecological effects characterization phase entailed a compilation and analysis of the stressor-response relationships and any other evidence of adverse impacts from exposure to each herbicide. For the most part, available data consisted of the toxicity studies conducted in support of USEPA pesticide registration described in Section 3.1. TRVs selected for use in the ERA are presented in Table 3-1. Appendix A presents the full set of toxicity information identified for fluridone.

In order to address potential risks to ecological receptors, RQs were calculated by dividing the EEC for each of the previously described scenarios by the appropriate TRV presented in Table 3-1. An RQ was calculated by dividing the EEC for a particular scenario by an herbicide specific TRV. The TRV may be a surface water or surface soil effects concentration, or a species-specific toxicity value derived from the literature.

The RQs were then compared to LOCs established by the USEPA OPP to assess potential risk to non-target organisms. Table 4-1 presents the LOCs established for this assessment. Distinct USEPA LOCs are currently defined for the following risk presumption categories:

- **Acute high risk -** the potential for acute risk is high.
- **Acute restricted use -** the potential for acute risk is high, but may be mitigated through a restricted use designation.
- **Acute endangered species** – the potential for acute risk to endangered species is high.
- **Chronic risk -** the potential for chronic risk is high.

Additional uncertainty factors may also be applied to the standard LOCs to reflect uncertainties inherent in extrapolating from surrogate species toxicity data to obtain RQs (see Sections 6.3 and 7.0 for a discussion of uncertainty). A "chronic endangered species" risk presumption category for aquatic animals was added for this risk assessment. The LOC for this category was set to 0.5 to reflect the conservative two-fold difference in contaminant sensitivity between RTE and surrogate test fishes (Sappington et al. 2001). Risk quotients predicted for acute scenarios (e.g., direct spray, accidental spill) were compared to the three acute LOCs, and the RQs predicted for chronic scenarios (e.g., long term ingestion) were compared to the two chronic LOCs. If all RQs were less than the most conservative LOC for a particular receptor, comparisons against other, more elevated LOCs were not necessary.

The RQ approach used in this ERA provides a conservative measure of the potential for risk based on a "snapshot" of environmental conditions (i.e., rainfall, slope) and receptor assumptions (i.e., BW, ingestion rates). Sections 6.3 and 7.0 discuss several of the uncertainties inherent in the RQ methodology.

To specifically address potential impacts to RTE species, two types of RQ evaluations were conducted. For RTE terrestrial plant species, the RQ was calculated using different toxicity endpoints but keeping the same LOC (set at 1) for all scenarios. The plant toxicity endpoints were selected to provide extra protection to the RTE species. In the direct spray and spray drift scenarios, the selected toxicity endpoints were an effect concentration (EC_{25}) for "typical" species and a NOAEL for RTE species. Potential impacts to non-target terrestrial plants from fluridone could not be evaluated quantitatively due to a lack of terrestrial plant toxicity data.

The evaluation of RTE terrestrial wildlife and aquatic species is addressed using a second type of RQ evaluation. The same toxicity endpoint was used for both typical and RTE species in all scenarios, but the LOC was lowered for RTE species.

4.3 Risk Characterization

The ecological risk characterization integrates the results of the exposure and effects phases (i.e., risk analysis), and provides comprehensive estimates of actual or potential risks to ecological receptors. Risk quotients are summarized in Tables 4-2 to 4-3 and presented graphically in Figures 4-3 to 4-6. The results are discussed below for each of the evaluated exposure scenarios.

Box plots are used to graphically display the range of RQs obtained from evaluating each receptor and exposure scenario combination (Figures 4-3 to 4-6). These plots illustrate how RQ data are distributed about the mean and their relative relationships with LOCs. Outliers (data points outside the 90^{th} or 10^{th} percentile) were not discarded in this ERA; all RQ data presented in these plots were included in the risk assessment.

4.3.1 Direct Spray

As described in Section 4.2.1, potential impacts from direct spray were evaluated for exposure that could occur within the aquatic application area (direct spray of pond during normal application, consumption of fish from contaminated pond) and outside the intended application area (accidental direct spray of terrestrial wildlife and non-target terrestrial plants, indirect contact with foliage, ingestion of contaminated prey items, accidental direct spray over stream). Table 4-2 presents the RQs for the following scenarios: direct spray of terrestrial wildlife, indirect contact with foliage after direct spray, ingestion of contaminated prey items by terrestrial wildlife, direct spray of non-target terrestrial plants, and direct spray over a pond or stream. Figures 4-3 to 4-6 present graphic representations of the range of RQs and associated LOCs.

4.3.1.1 Terrestrial Wildlife

Acute RQs for terrestrial animals (Figure 4-3) were below the most conservative LOC of 0.1 (acute endangered species) for all scenarios. Only one chronic exposure scenario exceeded the terrestrial animal chronic LOC. At the maximum application rate, the small mammalian herbivore had an RQ of 2.22, all other RQs were well below the LOC of 1.These results indicate that accidental direct spray impacts are not likely to pose a risk to insects, birds, or mammals under most conditions.

4.3.1.2 Non-target Plants – Terrestrial and Aquatic

No toxicity data was identified for non-target terrestrial plant species; therefore, a quantitative evaluation is not possible. However, the ecological incident report described in Section 2.3 suggests that impacts to terrestrial plants are possible due to unintended contact with fluridone. In the manufacturer's user's guide for the Sonar aquatic herbicide (Eli Lilly and Company 2003), grasses and some sedges are considered to be "sensitive" or "intermediate" in their tolerance to the herbicide, while rushes tend to be "intermediate" to "tolerant". Shoreline plants, such as willow and cypress, were considered "tolerant," while the tolerance of members of the evening primrose and acanthus families was classified as "intermediate." No concentrations were associated with these qualitative statements. The incident report and the user's guide both indicate that fluridone may cause negative impacts to terrestrial plants (e.g., tomatoes, grasses, sedges), but that shoreline plants are more tolerant. It is these more tolerant shoreline plants that are more likely to come in contact with fluridone during normal pond applications. The Sonar labels (SePRO 2002a,b,c; 2003) warn against using treated water for irrigation purposes for seven to thirty days after treatment. Even at the low fluridone concentrations used to treat milfoil, some terrestrial plants may be sensitive to fluridone if they are watered with treated lake water.

For aquatic plants, all of the RQs were below the plant LOC of 1, indicating that direct spray impacts are not predicted to pose a risk to aquatic plants in the stream or the pond. According to the Sonar user's guide (Eli Lilly and Company 2003), many native aquatic plants are tolerant to fluridone and show little or no impact following treatment. However, the target nuisance species, hydrilla, Eurasian watermilfoil, and curlyleaf pondweed, are highly susceptible to this herbicide.

4.3.1.3 Fish and Aquatic Invertebrates

Normal application of fluridone within a pond resulted in one RQ elevated over the associated LOC. The acute RQ for aquatic invertebrates in the pond impacted by the maximum application rate of fluridone was 0.11, just above the LOC for acute risk to endangered species (0.05). However, this value is below the acute high risk LOC, suggesting minimal risk to non-endangered species.

Accidental direct spray of fluridone over the stream results in elevated acute and chronic RQs (Figure 4-5 and 4-6). Elevated acute RQs were 0.17 for fish at the maximum application rate, and 0.065 and 0.56 for invertebrates at the typical and maximum application rates, respectively. These RQs were all above the acute risk to endangered species LOC, but below or nearly consistent with the acute high risk LOC. Elevated chronic RQs were 1.5 for fish and 1.8 for invertebrates at the maximum application rate. These RQs were above the LOC for chronic risk to endangered species (0.5) and the LOC for chronic risk (1).

These results indicate there is potential for risk to aquatic species, especially endangered species, in a stream sprayed with fluridone. It may be noted that these spray scenarios are very conservative because they are instantaneous concentrations and do not consider flow, adsorption to particles, or degradation that may occur over time. In addition, this scenario is not likely to occur as fluridone is reserved for use in ponds.

4.3.1.4 Piscivorous Birds

Risk to piscivorous birds (Figure 4-3) was assessed by evaluating impacts from consumption of fish from a pond impacted by normal application of fluridone. RQs for the piscivorous bird were all well below the most conservative terrestrial animal LOC (0.1), indicating that this scenario is not likely to pose a risk to piscivorous birds.

4.3.2 Off-site Drift to Non-target Terrestrial Plants

As described in Section 4.2.1, AgDRIFT® software was used to evaluate a number of possible scenarios in which a portion of the applied herbicide drifts outside of the treatment area and deposits onto non-target receptors. Ground applications of fluridone were modeled using both a low- and high-placed boom (spray boom height set at 20 and 50 inches above the ground, respectively), and aerial applications were modeled from both a helicopter and a plane over non-forested lands. Drift deposition was modeled at 25, 100, and 900 ft from the application area for ground applications and 100, 300, and 900 ft from the application aerial applications area.

As described previously, no toxicity data was identified for non-target terrestrial plant species, therefore a quantitative evaluation of this scenario is not possible. However, the ecological incident report described in Section 2.3 suggests that impacts to terrestrial plants are possible due to unintended contact with fluridone. As described in Section 4.3.1.2, the Sonar user's guide (Eli Lilly and Company 2003) and labels (SePRO, 2002a,b,c; SePRO 2003) indicate the potential for impact to non-target terrestrial plants.

It may be noted that the concentrations of fluridone predicted due to off-site drift are significantly lower than those modeled for accidental direct spray of fluridone on near shore terrestrial plants. Table 4-3 presents the soil deposition predicted as a result of off-site drift compared to herbicide concentrations resulting from the typical and maximum application rates considered in the direct spray scenarios (Section 4.3.1.2). This comparison indicates that the maximum deposition (100 ft from aerial applications) was only 23.8% of the typical application rate and only 0.87% of the maximum application rate. In general, off-site drift modeled using the typical application rate was < 10% of the typical application rate used in the direct spray scenario. Off-site drift modeled using the maximum application rate was < 1% of the maximum application rate used in the direct spray scenario. This table indicates the significant reduction in deposition and associated risks that occurs with off-site drift relative to direct accidental spray. It may be noted that a significantly greater proportion of the herbicide is deposited due to drift from aerial applications than from ground applications.

4.3.3 Accidental Spill to Pond

As described in Section 4.2.1, two spill scenarios were considered. These consist of a truck and a helicopter spilling entire loads (200 gal spill and 140 gal spill, respectively) of herbicide mixed for the maximum application rate into the 1/4 acre, 1 meter deep pond. The herbicide concentration in the pond was the instantaneous concentration at the moment of the spill. The volume of the pond was determined and the volume of herbicide in the truck and helicopter, respectively, were mixed into the pond volume.

Risk quotients for the truck spill scenario (Table 4-2) were 1.10 for fish, 3.58 for aquatic invertebrates (Figure 4-5 and 4-6), and 1.56 for non-target aquatic plants (Figure 4-4). Risk quotients for the helicopter spill scenario were slightly higher at 3.83, 12.6, and 5.44 for fish, aquatic invertebrates, and non-target aquatic plants, respectively. These scenarios are highly conservative and represent unlikely and worst case conditions (limited waterbody volume, tank mixed for maximum application). Spills of this magnitude are possible, but are not likely to occur. However, potential risks to fish, aquatic invertebrates, and non-target aquatic plants were indicated for the truck and helicopter spills mixed for the maximum application rate.

4.3.4 Potential Risk to Salmonids from Indirect Effects

In addition to direct effects of herbicides on salmonids and other fish species in stream habitats (i.e., mortality due to herbicide concentrations in surface water), reduction in vegetative cover or food supply may indirectly impact individuals or populations. No literature studies were identified that explicitly evaluated the indirect effects of fluridone to salmonids and their habitat; therefore, only qualitative estimates of indirect effects are possible. These estimates were accomplished by discussing predicted impacts to prey items and vegetative cover in the accidental direct spray over the stream scenario evaluated above. The only stream evaluation conducted for this risk assessment was the accidental direct spray scenario, since fluridone is not proposed for use in streams. An evaluation of impacts to non-target terrestrial plants was also included as part of the discussion of vegetative cover within the riparian zone. Prey items for salmonids and other potential RTE species may include other fish species, aquatic invertebrates, or aquatic plants. Additional discussion of RTE species is provided in Section 6.0.

4.3.4.1 Qualitative Evaluation of Impacts to Prey

Fish species were evaluated directly in the ERA using acute and chronic TRVs based on the most sensitive warm- or coldwater species identified during the literature search. Several laboratory studies with salmonids (rainbow trout) were identified in the literature and considered in the selection of the fish TRVs (Appendix A). The chronic fish TRV was based on a warm-water species, the fathead minnow. The acute fish TRV was based the rainbow trout, a salmonid. The inclusion of salmonid data in the TRV derivation reduced the uncertainties inherent in assessing potential indirect impacts to salmonids.

Aquatic invertebrates were also evaluated directly using acute and chronic TRVs based on the most sensitive aquatic invertebrate species. RQs in excess of the acute LOCs for fish and aquatic invertebrates were observed for the accidental direct spray scenario. However, this is an extremely conservative scenario in which it is assumed that a stream is accidentally directly sprayed by an aquatic herbicide intended for a pond. This is unlikely to occur as a result of BLM practices and represents a worst-case scenario. In addition, stream flow would be likely to dilute the herbicide concentration and reduce potential impacts, but no reduction in herbicide concentration is calculated as a result of stream flow.

The only stream evaluation conducted for this risk assessment was an accidental direct spray scenario and may overestimate risk to aquatic stream receptors. However, this conservative evaluation predicts that fish and aquatic invertebrates may be directly impacted by herbicide concentrations in the stream. Accordingly, their availability as prey item populations may be impacted and there may be an indirect effect on salmonids.

4.3.4.2 Qualitative Evaluation of Impacts to Vegetative Cover

A qualitative evaluation of indirect impacts to salmonids due to destruction of riparian vegetation and reduction of available cover was made by considering impacts to terrestrial and aquatic plants. Aquatic plant RQs for accidental direct spray scenarios were below the plant LOC at both the typical and maximum application rates, indicating that impacts to the aquatic plant community are not predicted. This evaluation indicates that indirect impacts to salmonids due to a reduction in available cover are unlikely.

Although terrestrial plants were not specifically evaluated in the stream scenarios of the ERA, a reduction in riparian cover has the potential to indirectly impact salmonids within the stream. However, terrestrial plant TRVs were not available for this evaluation. A review of incident reports and the manufacturer's user's guide (Eli Lilly and Company 2003) indicate that shoreline plant species are generally tolerant of fluridone exposures. However, the user's guide (Eli Lilly and Company 2003) and labels (SePRO, 2002a,b,c; SePRO 2003) do indicate the potential for impact to non-target terrestrial plants. Therefore, it is uncertain whether or not a reduction in riparian cover is likely.

4.3.4.3 Conclusions

This qualitative evaluation indicates that salmonids may be indirectly impacted by a reduction in food supply (i.e., fish and aquatic invertebrates). However, this evaluation is based on worst-case accidental exposure scenarios that are not likely to occur as a result of BLM management practices. Reducing the application rate and avoidance of accidental application on non-target areas would reduce the likelihood of these impacts. A reduction in aquatic vegetative cover was not predicted. Based on a lack of toxicity data, it is unknown whether a reduction in terrestrial plant cover would occur.

In addition, the effects of aquatic herbicides in water are expected to be relatively transient and stream flow is likely to reduce herbicide concentrations over time. Only very persistent pesticides would be expected to have effects beyond the year of their application. An OPP report on the impacts of a terrestrial herbicide on salmonids indicated that if a listed salmonid was not present during the year of application, there would likely be no concern (Turner 2003). Therefore, it is expected that potential adverse impacts to food and aquatic cover would not be occur beyond the season of application.

TABLE 4-1
Levels of Concern

	Risk Presumption	RQ	LOC
Terrestrial Animals [1]			
Birds	Acute High Risk	EEC/LC_{50}	0.5
	Acute Restricted Use	EEC/LC_{50}	0.2
	Acute Endangered Species	EEC/LC_{50}	0.1
	Chronic Risk	EEC/NOAEL	1
Wild Mammals	Acute High Risk	EEC/LC_{50}	0.5
	Acute Restricted Use	EEC/LC_{50}	0.2
	Acute Endangered Species	EEC/LC_{50}	0.1
	Chronic Risk	EEC/NOAEL	1
Aquatic Animals [2]			
Fish and Aquatic Invertebrates	Acute High Risk	EEC/LC_{50} or EC_{50}	0.5
	Acute Restricted Use	EEC/LC_{50} or EC_{50}	0.1
	Acute Endangered Species	EEC/LC_{50} or EC_{50}	0.05
	Chronic Risk	EEC/NOAEL	1
	Chronic Risk, Endangered Species	EEC/NOAEL	0.5
Plants [3]			
Terrestrial Plants	Acute High Risk	EEC/EC_{25}	1
	Acute Endangered Species	EEC/NOAEL	1
Aquatic Plants	Acute High Risk	EEC/EC_{50}	1
	Acute Endangered Species	EEC/NOAEL	1

[1] Estimated Environmental Concentration (EEC) is in mg $_{prey}$/kg $_{body\ weight}$ for acute scenarios and mg $_{prey}$/kg $_{body\ weight}$/day for chronic scenarios.
[2] EEC is in mg/L.
[3] EEC is in lbs/ac.

TABLE 4-2
Risk Quotients for Direct Spray and Spill Scenarios

Terrestrial Animals	Typical Application Rate	Maximum Application Rate
Direct Spray of Terrestrial Wildlife		
Small mammal - 100% absorption	1.95E-04	1.69E-03
Pollinating insect - 100% absorption	2.03E-03	1.76E-02
Small mammal - 1st order dermal adsorption	5.54E-06	4.80E-05
Indirect Contact With Foliage After Direct Spray		
Small mammal - 100% absorption	1.95E-05	1.69E-04
Pollinating insect - 100% absorption	2.03E-04	1.76E-03
Small mammal - 1st order dermal adsorption	5.54E-07	4.80E-06
Ingestion of Prey Items Contaminated by Direct Spray		
Small mammalian herbivore - acute exposure	2.90E-05	1.89E-03
Small mammalian herbivore - chronic exposure	3.40E-02	**2.22E+00**
Large mammalian herbivore - acute exposure	1.86E-04	8.81E-03
Large mammalian herbivore - chronic exposure	9.27E-03	4.40E-01
Small avian insectivore - acute exposure	2.33E-04	1.57E-02
Small avian insectivore - chronic exposure	4.66E-03	3.14E-01
Large avian herbivore - acute exposure	5.67E-04	4.16E-02
Large avian herbivore - chronic exposure	1.18E-02	8.68E-01
Large mammalian carnivore - acute exposure	1.21E-04	1.05E-03
Large mammalian carnivore - chronic exposure	1.87E-04	1.62E-03

Semi-Aquatic Wildlife	Typical Application Rate	Maximum Application Rate
Ingestion of Prey Items Contaminated by Normal Application to Pond		
Avian piscivore – chronic exposure	4.00E-05	3.47E-04

TABLE 4-2 (Cont.)
Risk Quotients for Direct Spray and Spill Scenarios

Terrestrial Plants	Typical Species		Rare, Threatened, and Endangered Species	
	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate
Direct Spray of Non-Target Terrestrial Plants				
Accidental direct spray	NC	NC	NC	NC

Aquatic Species	Fish		Aquatic Invertebrates		Non-Target Aquatic Plants	
	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate
Direct Spray Over Pond – Normal Application						
Acute	3.96E-03	3.43E-02	1.29E-02	**1.12E-01**	5.60E-03	4.86E-02
Chronic	3.36E-02	2.91E-01	2.80E-02	2.43E-01	1.68E-02	1.46E-01
Direct Spray Over Stream – Accidental Spray						
Acute	1.98E-02	**1.71E-01**	**6.47E-02**	**5.60E-01**	2.80E-02	2.43E-01
Chronic	1.68E-01	**1.46E+00**	1.40E-01	**1.21E+00**	8.41E-02	7.29E-01
Accidental spill						
Truck spill into pond	--	**1.10E+00**	--	**3.59E+00**	--	**1.55E+00**
Helicopter spill into pond	--	**3.84E+00**	--	**1.26E+01**	--	**5.44E+00**

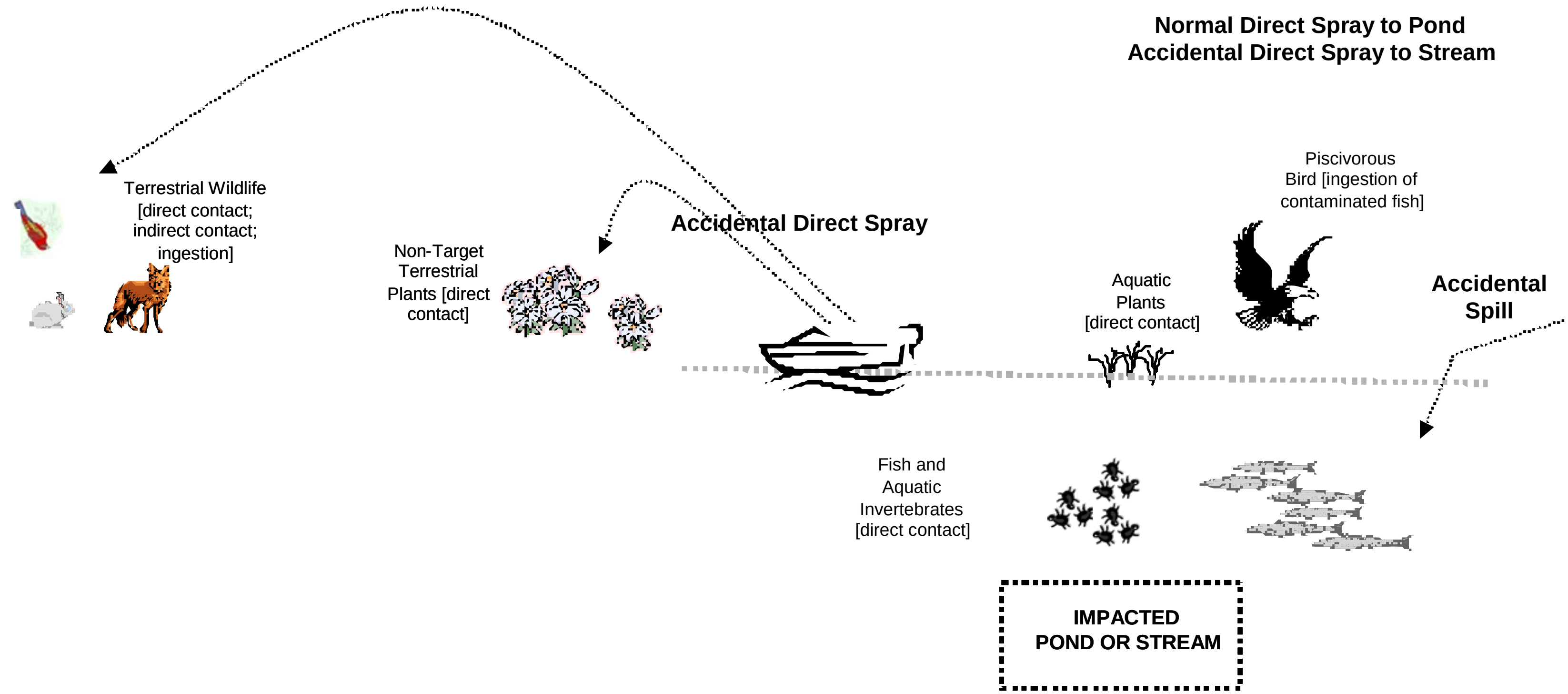
NC - Not calculated. RQs could not be calculated due to a lack of terrestrial plant toxicity testing. Only a qualitative evaluation was possible.
Shading and boldface indicates terrestrial animal acute RQs greater than 0.1 (LOC for acute risk to endangered species - most conservative).
Shading and boldface indicates terrestrial animal chronic RQs greater than 1 (LOC for chronic risk).
Shading and boldface indicates plant RQs greater than 1 (LOC for all plant risks).
Shading and boldface indicates acute RQs greater than 0.05 for fish and invertebrates (LOC for acute risk to endangered species - most conservative).
Shading and boldface indicates chronic RQs greater than 0.5 for fish and invertebrates (LOC for chronic risk to endangered species).
RTE – Rare, threatened, and endangered.
-- indicates the scenario was not evaluated

TABLE 4-3
Comparison of Soil Deposition Due to Off-Site Drift and Direct Spray

Soil Deposition						
			Typical Application Rate		Maximum Application Rate	
Mode of Application	**Application Height or Type**	**Distance From Receptor (ft)**	**lbs a.i./ac**	**%**	**lbs a.i./ac**	**%**
OFF-SITE DRIFT (modeled in AgDRIFT)						
Plane	Non-Forested	100	3.57E-02	[23.8]	1.13E-02	[0.87]
Plane	Non-Forested	300	1.78E-02	[11.9]	5.94E-03	[0.46]
Plane	Non-Forested	900	5.92E-03	[3.94]	2.80E-03	[0.22]
Helicopter	Non-Forested	100	3.57E-02	[23.8]	9.42E-03	[0.72]
Helicopter	Non-Forested	300	8.92E-03	[5.95]	4.62E-03	[0.36]
Helicopter	Non-Forested	900	4.75E-03	[3.16]	2.01E-03	[0.15]
Ground	Low Boom	25	5.15E-03	[3.43]	9.13E-04	[0.07]
Ground	Low Boom	100	1.82E-03	[1.21]	5.01E-04	[0.039]
Ground	Low Boom	900	2.79E-04	[0.19]	9.67E-05	[0.007]
Ground	High Boom	25	8.51E-03	[5.67]	1.47E-03	[0.11]
Ground	High Boom	100	2.86E-03	[1.91]	7.73E-04	[0.059]
Ground	High Boom	900	3.58E-04	[0.24]	1.23E-04	[0.009]
DIRECT SPRAY						
			1.50E-01		1.30E+00	

Value in brackets indicates percentage of the direct spray application rate that is deposited due to off-site drift.

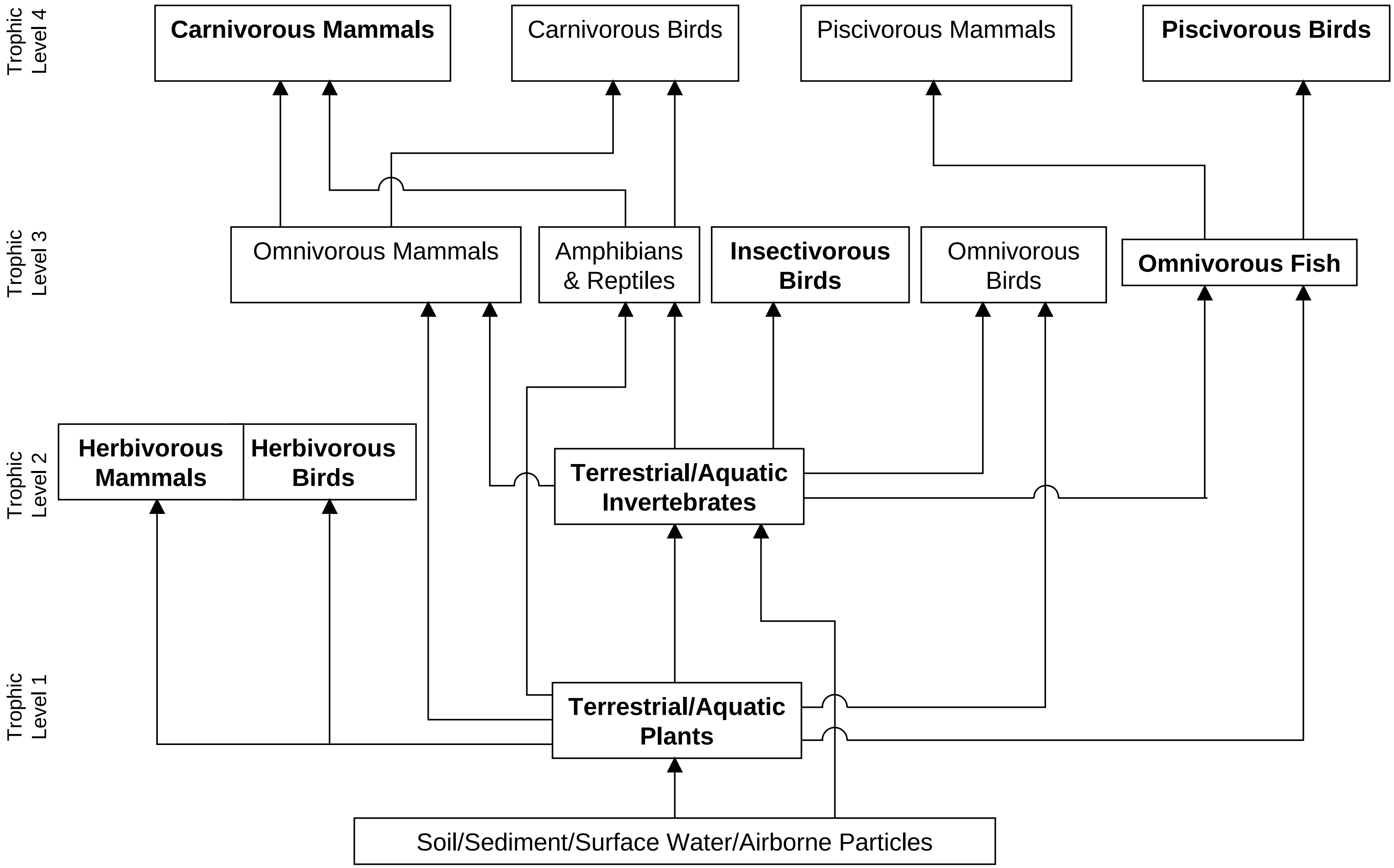
FIGURE 4-1. Conceptual Model for Aquatic Herbicides.



Application of aquatic herbicides may occur by aerial (i.e., plane, helicopter) ground (i.e., truck, backpack), or boat methods.

See Figure 4-2 for simplified food web & evaluated receptors.

FIGURE 4-2. Simplified Food Web.



Receptors in **bold** type quantitatively assessed in the BLM herbicide ERAs.

Figure 4-3. Direct Spray - Risk Quotients for Terrestrial Animals & Semi-Aquatic Wildlife.

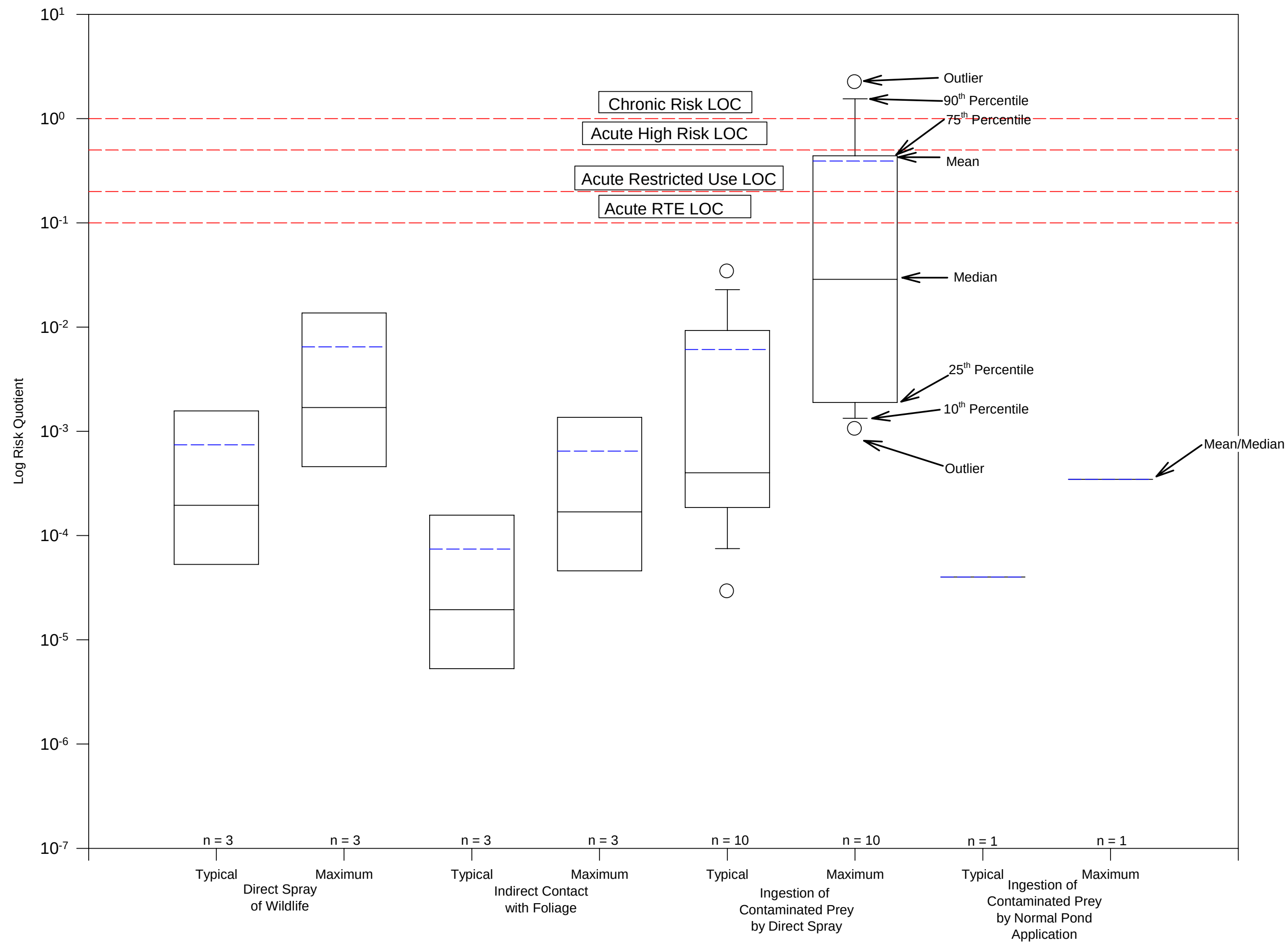


FIGURE 4-4. Accidental Direct Spray and Spills - Risk Quotients for Non-Target Aquatic Plants.

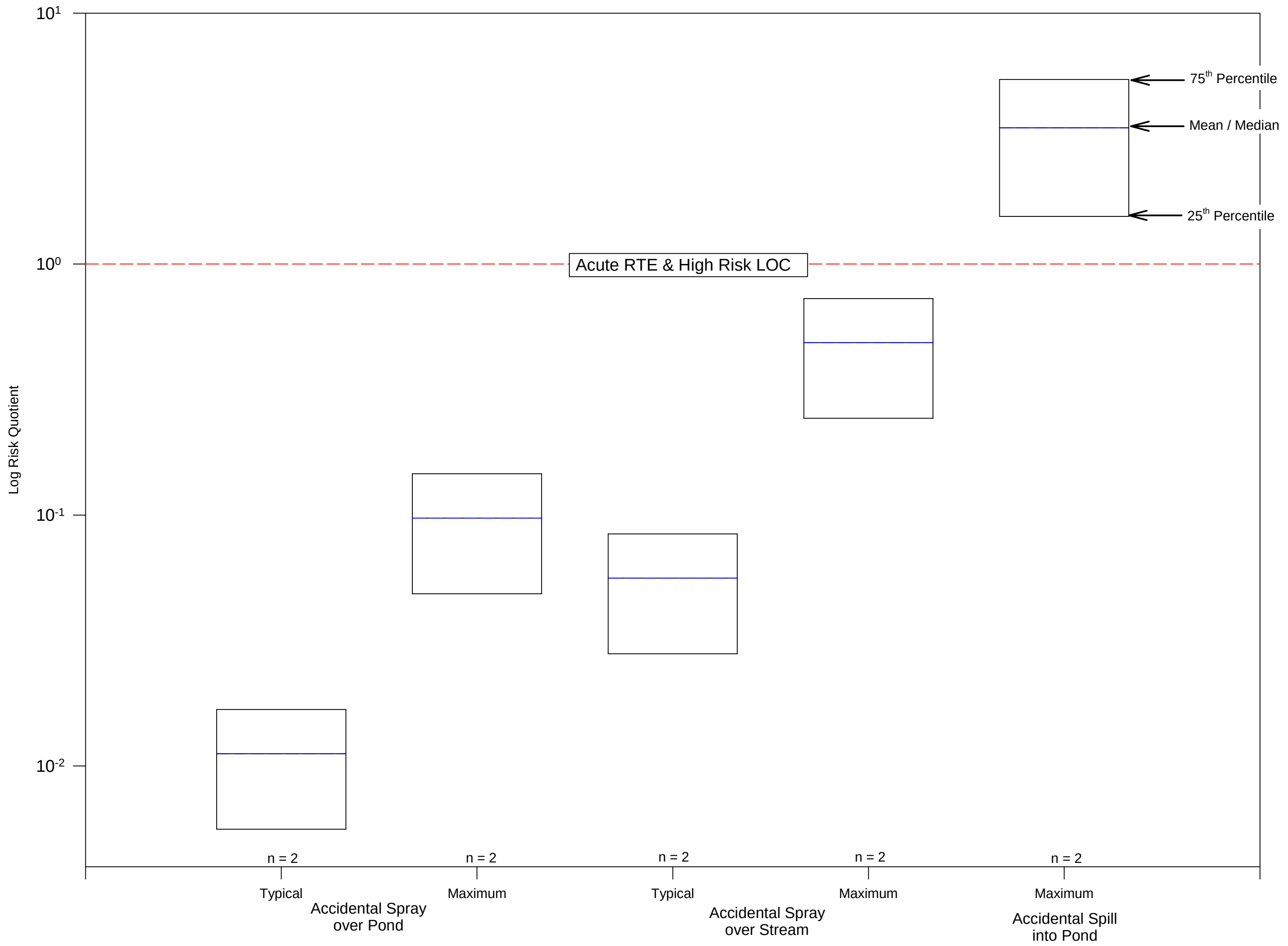


Figure 4-5. Accidental Direct Spray and Spills - Risk Quotients for Fish.

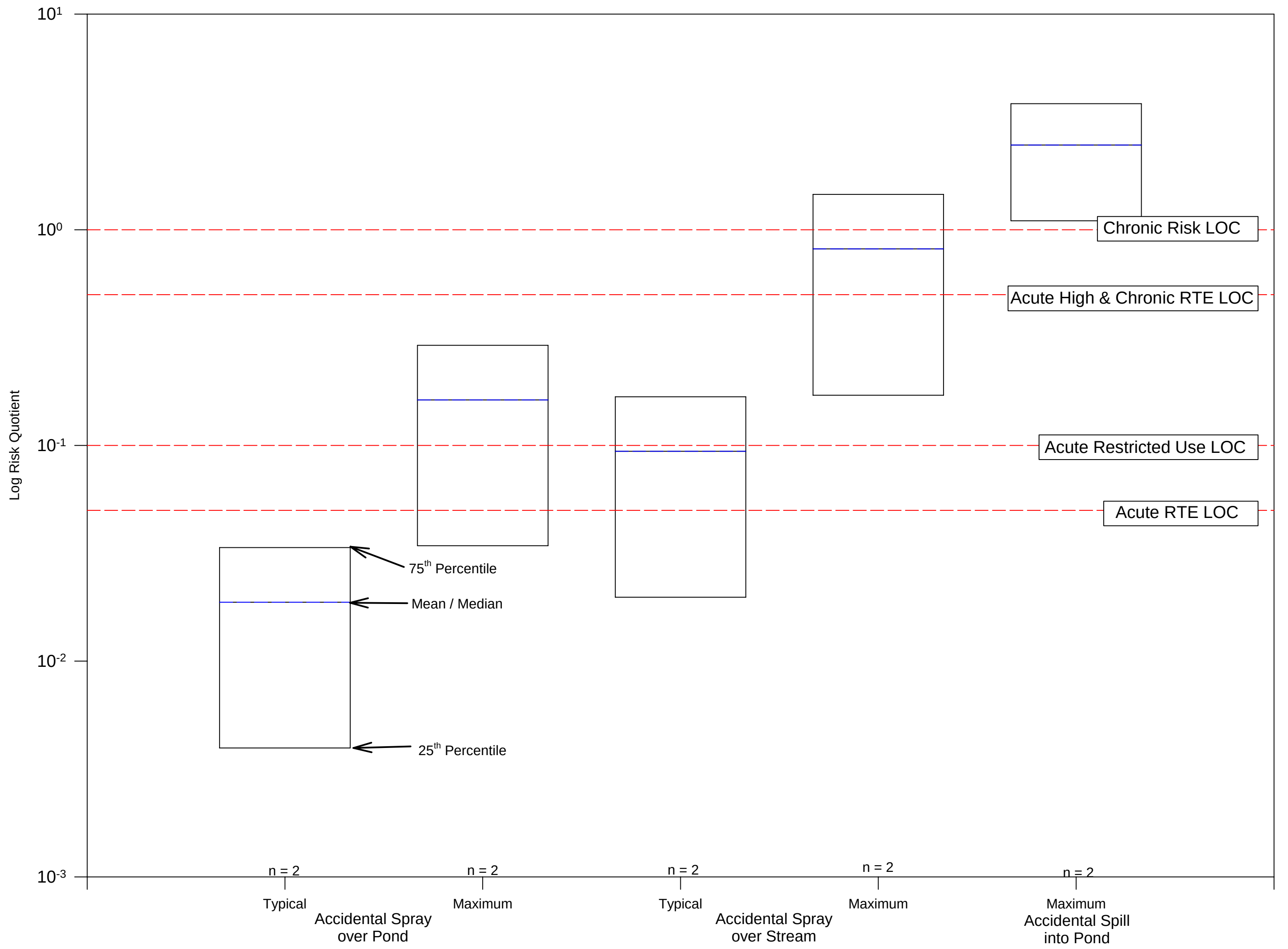
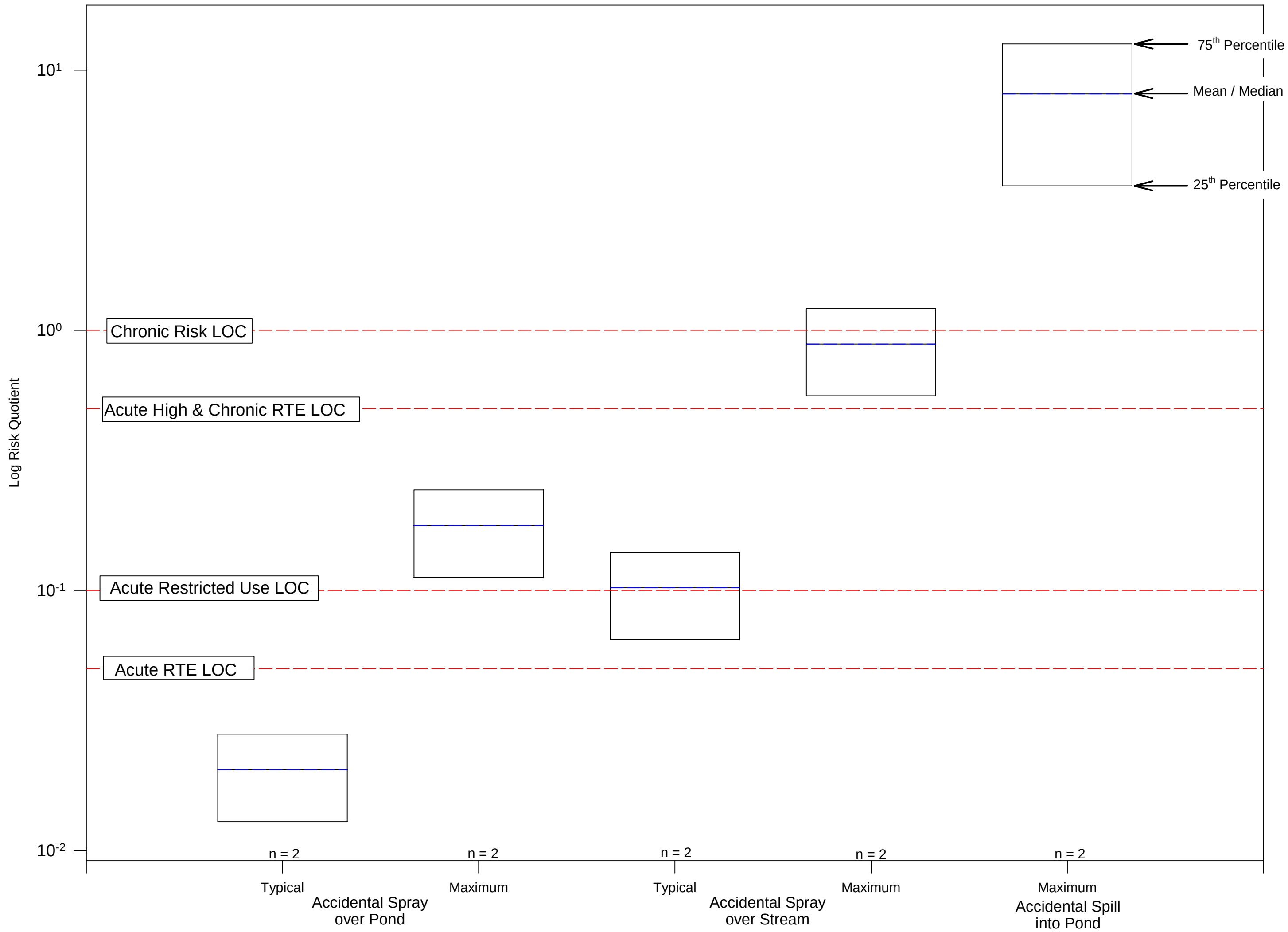


Figure 4-6. Accidental Direct Spray and Spills - Risk Quotients for Aquatic Invertebrates.



5.0 SENSITIVITY ANALYSIS

The sensitivity analysis was designed to determine which factors most greatly affect exposure concentrations. Changes in herbicide concentrations were modeled with respect to changes in pond and stream area and depth. The effects of off-site drift on terrestrial species were estimated using the AgDRIFT® model. A base case for the AgDRIFT® model was established, and from this base case various input factors were changed independently, thereby resulting in an estimate of the importance of that factor on exposure concentrations. Information regarding the AgDRIFT® model, its specific use and any inputs and assumptions made during the application of this model is provided in the Methods Document (ENSR 2004c).

5.1 Pond Volume and Stream Flow Sensitivity

The sensitivity analysis was designed to determine how pond and stream volumes affect exposure concentrations. A base case for each model was established. Input factors (e.g., area, depth) were changed independently, thereby resulting in an estimate of the importance of that factor on exposure concentrations. As described previously, surface runoff and wind erosion were not considered as transport mechanisms for the aquatic herbicides. The scenarios for the aquatic herbicides are relatively simplistic and essentially represent an instantaneous concentration in the waterbody due to direct applications. The predicted surface water concentrations are based on the application rate, and the surface area and depth of the waterbody. The surface water concentrations predicted in these scenarios are likely to be an overestimate since stream flow, degradation, and adsorption are not considered.

The base case for the pond consisted of a ¼ acre pond 1 meter deep. Table 5-1 presents the variations in the pond surface water concentrations as the area and depth of the pond are changed. This analysis indicates that changing the area of the pond does not alter the predicted surface water concentration because as more herbicide is sprayed over a larger area, there is a larger pond volume in which the herbicide is dissipated. However, changing the depth does have an impact on the pond concentration because the pond volume changes, but the amount of herbicide sprayed on the pond is unchanged. For example, an increase in the pond depth will decrease the associated herbicide concentration in the surface water.

The base case for the stream consisted of a stream 2 m wide and 0.2 m deep. The base case length was based on one side of a 100 acre square application area (636 m). Table 5-2 presents the variations in the stream surface water concentrations as the width, length, and depth of the impacted stream are changed. As observed in the pond sensitivity analysis, changes to stream area accomplished by varying the length or width do not result in changes to the surface water concentrations. Changes to the stream depth do result in associated changes to the stream concentrations. As the depth is increased, the stream concentration decreases and as the depth decreases, the stream concentration increases.

The results of this sensitivity analysis indicate that the size of the impacted water body does not have an effect on the surface water concentration (assuming that the entire waterbody is sprayed). However, depth has a dramatic impact on the associated surface water concentration (doubling the depth decreased the water concentration by ½). This indicates that shallow ponds and streams are more likely to be impacted by herbicide spray.

5.2 AgDRIFT® Sensitivity

Changes to individual input parameters of predictive models have the potential to substantially influence the results of an analysis such as that conducted in this ERA. This is particularly true for models such as AgDRIFT® which are intended to represent complex problems such as the prediction of off-target spray drift of herbicides. Predicted off-target spray drift and downwind deposition can be substantially altered by a number of variables intended to represent the herbicide application process including, but not limited to: nozzle type used in the spray application of an herbicide mixture, ambient wind speed, release height (application boom height), and evaporation. Hypothetically, any variable in the model that is intended to represent some part of the physical process of spray drift and deposition can substantially alter predicted downwind drift and deposition patterns. This section will present the changes that occur to the EEC with changes to important input parameters and assumptions used in the AgDRIFT® model. It is important to note that changes in the EEC directly affect the estimated RQ. Thus, this information is presented to help local land managers understand the factors that are likely to be related to higher potential ecological risk. Table 5-3 summarizes the relative change in exposure concentrations, and therefore ecological risk, based on specific model input parameters (e.g., mode of application, application rate).

Factors that are thought to have the greatest influence on downwind drift and deposition are: spray drop-size distribution, release height, and wind speed (Teske and Barry 1993; Teske et al. 1998; Teske and Thistle 1999, *as cited in SDTF 2002*). To better quantify the influence of these and other parameters, a sensitivity analysis was undertaken by the SDTF and documented in the AgDRIFT® user's manual. In this analysis AgDRIFT® Tier II model input parameters (model input parameters are discussed in Appendix B of the HHRA) were varied by 10% above and below the default assumptions (four different drop-size distributions were evaluated). The findings of this analysis indicate the following:

- The largest variation in predicted downwind drift and deposition patterns occurred as a result of changes in the shape and content of the spray drop size distribution.
- The next greatest change in predicted downwind drift and deposition patterns occurred as a result of changes in boom height (the release height of the spray mixture).
- Changes in spray boom length resulted in significant variations in drift and deposition within 200 ft downwind of the hypothetical application area.
- Changes in the assumed ambient temperature and relative humidity resulted in small variation in drift and deposition at distances > 200 ft downwind of the hypothetical application area.
- Varying the assumed number of application swaths (aircraft flight lines), application swath width, and wind speed resulted in little change in predicted downwind drift and deposition.
- Variation in nonvolatile fraction of the spray mixture showed no effect on downwind drift and deposition.

These results, except for the minor to negligible influence of varying wind speed and nonvolatile fraction, were consistent with previous observations. The 10% variation in wind speed and nonvolatile fraction was likely too small to produce substantial changes in downwind drift and deposition. It is expected that varying these by a larger percentage would eventually produce some effect. In addition, changes in wind speed resulted in changes in application swath width and swath offset, which masked the effect of wind speed alone on downwind drift and deposition.

Based on these findings, and historic field observations, the hierarchy of parameters that have the greatest influence on downwind drift and deposition patterns is as follows:

1. Spray drop size distribution
2. Application boom height
3. Wind speed
4. Spray boom length
5. Relative humidity
6. Ambient temperature
7. Nonvolatile fraction

An additional limitation of the AgDRIFT® user's manual sensitivity analysis is the focus on distances < 200 ft downwind of a hypothetical application area. From a land management perspective, distance downwind from the point of deposition may be considered to represent a hypothetical buffer zone between the application area and a potentially sensitive habitat. In this ERA, distances as great as 900 ft downwind of a hypothetical application were considered. In an effort to expand on the existing AgDRIFT® sensitivity analysis provided in the user's manual, the sensitivity of mode of application, application height or vegetation type, and application rate were evaluated. Results of this supplemental analysis are provided in Table 5-3.

The results of the expanded sensitivity analysis indicate that deposition and corresponding ecological risk drop off substantially between 25 and 900 ft downwind of hypothetical application area. Thus, from a land management perspective, the size of a hypothetical buffer zone (the downwind distance from a hypothetical application area to a potentially sensitive habitat) may be the single most controllable variable (other than the application rate, equipment and herbicide mixtures chosen) that has a substantial impact on ecological risk (Table 5-3).

The most conservative case at the typical application rate (using the smallest downwind distance measured in this ERA – 25 ft) was then evaluated using two different boom heights. Predicted concentrations were greater with high vs. low boom height (Table 5-3); ecological risk, therefore, increases with boom height. The effect of mode of application was evaluated using plane, helicopter and ground dispersal (using the typical application rate, smallest downwind distance, and non-forested cover or high boom height). Plane dispersal resulted in the highest predicted exposure concentrations, and therefore, represents the greatest risk. Ground applications resulted in the lowest predicted exposure concentrations. The effect of application rate (maximum vs. typical) was also tested, and as expected, predicted concentrations (and ecological risk) increase with increased application rates (Table 5-3). Concentrations were approximately four times greater using maximum application rates than using typical application rates.

TABLE 5-1
Relative Effects of Pond Variables on Herbicide Exposure Concentrations using Typical BLM Application Rate

Pond area (acres)	Pond depth (m)	Pond volume (L)	Mass sprayed on pond (mg)	Concentration in pond (mg/L)	Comments
0.25	1	1,011,714	17,010	0.02	Base case
100	1	404,685,642	6,803,886	0.02	Increased pond area; No change in concentration
1000	1	4,046,856,422	68,038,856	0.02	Increased pond area; No change in concentration
0.25	0.5	2,023,428,211	17,010	0.03	Decreased pond depth; Increased concentration
0.25	2	2,023,428	17,010	0.008	Increased pond depth; Decreased concentration
0.25	4	4,046,856	17,010	0.004	Increased pond depth; Decreased concentration

TABLE 5-2
Relative Effects of Stream Variables on Herbicide Exposure Concentrations using Typical BLM Application Rate

Stream width (m)	Stream depth (m)	Length of impacted stream (m) [1]	Stream volume (L)	Mass sprayed on stream (mg)	Concentration in stream (mg/L)	Comments
2	0.2	636	254,460	21,391	0.08	Base case
4	0.2	636	508,920	42,782	0.08	Increased stream width; No change in concentration
1	0.2	636	127,230	10,695	0.08	Decreased stream width; No change in concentration
2	0.4	636	508,920	21,391	0.04	Increased stream depth; Decreased concentration
2	0.1	636	127,230	21,391	0.17	Decreased stream depth; Increased concentration
2	0.2	201	80,468	6,764	0.08	Increased stream length; No change in concentration
2	0.2	2,012	804,672	67,644	0.08	Decreased stream length; No change in concentration

(1) – Length of impacted stream is based on size of application area. 10 acre application area = 201 meters impacted; 100 acre application area = 636 meters impacted; 1,000 acre application area = 2,012 meters impacted.

TABLE 5-3
Herbicide Exposure Concentrations Used During the Supplemental AgDRIFT® Sensitivity Analysis

Mode of Application	Application Height or Vegetation Type	Minimum Downwind Distance	Maximum Downwind Distance	Minimum Downwind Distance Concentration Pond (mg/L)	Maximum Downwind Distance Concentration Pond (mg/L)
Typical Application Rate					
Plane	Forest	100	900	NA	NA
	Non-Forest	100	900	2.94E-03	6.31E-04
Helicopter	Forest	100	900	NA	NA
	Non-Forest	100	900	2.50E-03	5.15E-04
Ground	Low Boom	25	900	2.79E-04	2.96E-05
	High Boom	25	900	4.49E-04	3.76E-05
Maximum Application Rate					
Plane	Forest	100	900	NA	NA
	Non-Forest	100	900	1.13E-02	2.80E-03
Helicopter	Forest	100	900	NA	NA
	Non-Forest	100	900	9.42E-03	2.01E-03
Ground	Low Boom	25	900	9.13E-04	9.67E-05
	High Boom	25	900	1.47E-03	1.23E-04

Effect of Downwind Distance

Mode of Application	Application Height or Vegetation Type	Minimum Downwind Distance	Maximum Downwind Distance	Concentration $_{900}$/ Concentration $_{25\ \text{or}\ 100}$	Relative Change in Concentration
Typical Application Rate					
Plane	Forest	100	900	NA	NA
	Non-Forest	100	900	0.2146	-
Helicopter	Forest	100	900	NA	NA
	Non-Forest	100	900	0.2060	-
Ground	Low Boom	25	900	0.1061	-
	High Boom	25	900	0.0837	-
Maximum Application Rate					
Plane	Forest	100	900	NA	NA
	Non-Forest	100	900	0.2478	-
Helicopter	Forest	100	900	NA	NA
	Non-Forest	100	900	0.2134	-
Ground	Low Boom	25	900	0.1059	-
	High Boom	25	900	0.0837	-

TABLE 5-3 (Cont.)
Herbicide Exposure Concentrations Used During the Supplemental AgDRIFT® Sensitivity Analysis

Effect of Application Vegetation Type or Boom Height

Mode of Application	Application Height or Vegetation Type	Vegetation Type or Boom Height[1]	Relative Change in Concentration
Typical Application Rate			
Plane	Forest/ Non-Forest	NA	NA
Helicopter	Forest/ Non-Forest	NA	NA
Ground	High/Low Boom	1.6093	+
Maximum Application Rate			
Plane	Forest/ Non-Forest	NA	NA
Helicopter	Forest/ Non-Forest	NA	NA
Ground	High/Low Boom	1.6101	+

Effect of Mode of Application

	Mode of Application[2]	Relative Difference
Typical Application Rate		
Plane vs. Helicopter	1.1760	+
Plane vs. Ground	6.5479	+
Helicopter vs. Ground	5.5679	+
Maximum Application Rate		
Plane vs. Helicopter	1.1996	+
Plane vs. Ground	7.6871	+
Helicopter vs. Ground	6.4082	+

Effect of Mode of Application Rate

	Application Rate[3]	Relative Difference
Maximum vs. Typical	3.2739	+

(1) using minimum buffer width concentrations.
(2) using minimum buffer width and non-forest or high boom concentrations.
(3) using ground dispersal, minimum buffer width, and high boom concentrations.
"+" = Increase in concentration = increase in RQ = increase in ecological risk.
"-" = Decrease in concentration = decrease in RQ = decrease in ecological risk.

6.0 RARE, THREATENED, AND ENDANGERED SPECIES

Rare, threatened, and endangered (RTE) species have the potential to be impacted by herbicides applied for vegetation control. RTE species are of potential increased concern to screening level ERAs, which utilize surrogate species and generic assessment endpoints to evaluate potential risk, rather than examining site- and species-specific effects to individual RTE species. Several factors complicate our ability to evaluate site- and species-specific effects:

- Toxicological data specific to the species (and sometimes even class) of organism are often absent from the literature.
- The other assumptions involved in the ERA (e.g., rate of food consumption, surface-to-volume ratio) may differ for RTE species relative to selected surrogates and/or data for RTE species may be unavailable.
- The high level of protection afforded RTE species by regulation and policy suggests that secondary effects (e.g., potential loss of prey or cover), as well as site-specific circumstances that might result in higher rates of exposure, should receive more attention.

A common response to these issues is to design screening level ERAs, including this one, to be highly conservative. This includes assumptions such as 100% exposure to an herbicide by simulating scenarios where the organism lives year-round in the most affected area (i.e., area of highest concentration), or that the organism consumes only food items that have been impacted by the herbicide. The fluridone screening level ERA incorporates additional conservatism in the assumptions used in the herbicide concentration models such as AgDRFIT® (Appendix A; ENSR 2004c). Even with highly conservative assumptions in the ERA, however, concern may still exist over the potential risk to specific RTE species.

To help address this potential concern, the following section will discuss the ERA assumptions as they relate to the protection of RTE species. The goals of this discussion are as follows:

- Present the methods the ERA employs to account for risks to RTE species and the reasons for their selection.
- Define the factors that might motivate a site- and/or species-specific evaluation[3] of potential herbicide impacts to RTE species and provide perspective useful for such an evaluation.
- Present information that is relevant to assessing the uncertainty in the conclusions reached by the ERA with respect to RTE species.

The following sections describe information used in the ERA to provide protection to RTE species, including mammals, birds, plants, reptiles, amphibians and fish (e.g., salmonids) potentially occurring on BLM-managed lands. It includes a discussion of the quantitative and qualitative factors used to provide additional protection to RTE species and a discussion of potential secondary effects of herbicide use on RTE species.

Section 6.1 provides a review of the selection of LOCs and TRVs with respect to providing additional protection to RTE species. Section 6.2 provides a discussion of species-specific traits and how they relate to the RTE protection strategy in this ERA. Section 6.2 also includes discussion of the selection of surrogate species (6.2.1), the RTE taxa of

[3] Such an evaluation might include site-specific estimation of exposure point concentrations using one or more models, more focused consideration of potential risk to individual RTE species; and/or more detailed assessment of indirect effects to RTE species, such as those resulting from impacts to habitat.

concern, and the surrogates used to represent them (6.2.2), and the biological factors that affect the exposure to and response of organisms to herbicides (6.2.3). This includes a discussion of how the ERA was defined to assure that consideration of these factors resulted in a conservative assessment. Mechanisms for extrapolating toxicity data from one taxon to another are briefly reviewed in Section 6.3. The potential for impacts, both direct and secondary, to salmonids is discussed in Section 6.4. Section 6.5 provides a summary of the section.

6.1 Use of LOCs and TRVs to Provide Protection

Potential direct impacts to receptors, including RTE species, are the measures of effect typically used in screening level ERAs. Direct impacts, such as those resulting from direct or indirect contact or ingestion were assessed in the fluridone ERA by comparing calculated RQs to receptor-specific LOCs. As described in the methodology document for this ERA (ENSR 2004c), RQs are calculated as the potential dose or EEC divided by the TRV selected for that pathway. An RQ greater than the LOC indicates the potential for risk to that receptor group via that exposure pathway. As described below, the selection of TRVs and the use of LOCs were pursued in a conservative fashion in order to provide a greater level of protection for RTE species.

The LOCs used in the ERA (Table 4-1) were developed by the USEPA for the assessment of pesticides (LOC information obtained from Michael Davy, USEPA OPP on 13 June 2002). In essence, the LOCs act as uncertainty factors often applied to TRVs. For example, using an LOC of 0.1 provides the same result as dividing the TRV by 10. The LOC for avian and mammalian RTE species is 1.0 for acute and chronic exposures. For RTE fish and aquatic invertebrates, acute and chronic LOCs were 0.05 and 0.5, respectively. Therefore, up to a 20-fold uncertainty factor has been included in the TRVs for animal species. As noted below, such uncertainty factors provide a greater level of protection to RTE species to account for the factors listed in the introduction to this section.

For RTE plants, the exposure concentration, TRVs, and LOCs provided a direct assessment of potential impacts. For all exposure scenarios, the maximum modeled concentrations were used as the exposure concentrations. The TRVs used for RTE plants were selected based on highly sensitive endpoints, such as germination, rather than direct mortality of seedlings or larger plants. Conservatism has been built into the TRVs during their development (Section 3.1); the lowest suitable endpoint concentration available was used as the TRV for RTE plant species. Therefore, the RQ calculated for RTE plant exposure is intrinsically conservative. Given the conservative nature of the RQ, and consistent with USEPA policy, no additional levels of protection were required for the LOC (all plant LOCs are 1).

6.2 Use of Species Traits to Provide Protection to RTE Species

Over 500 RTE species currently listed under the Federal Endangered Species Act (ESA) have the potential to occur in the 17 states covered under this Programmatic ERA. These species include 287 plants, 80 fish, 30 birds, 47 mammals, 15 reptiles, 13 amphibians, 34 insects, 10 arachnids (spiders), and 22 aquatic invertebrates (12 mollusks and 10 crustaceans).[4] Some marine mammals are included in the list of RTE species; but due to the limited possibility these species would be exposed to herbicides applied to BLM-managed lands, no surrogates specific to marine species are included in this ERA. However, the terrestrial mammalian surrogate species identified for use in the ERA include species that can be considered representative of these marine species as well. The complete list is presented in Appendix D.

Of the over 500 species potentially occurring in the 17 states, just over 300 species may occur on lands managed by the BLM. These species include 7 amphibians, 19 birds, 6 crustaceans, 65 fish, 30 mammals, 10 insects, 13 mollusks, 5 reptiles, and 151 plants. Protection of these species is an integral goal of the BLM, and they are the focus of the RTE evaluation for the ERA and EIS. These species are different from one another in regards to home range, foraging strategy, trophic level, metabolic rate, and other species-specific traits. Several methods were used in the ERA to take these differences into account during the quantification of potential risk. Despite this precaution, these traits are reviewed in order to provide a basis for potential site- and species-specific risk assessment. Review of these factors

[4] The number of RTE species may have changed slightly since the writing of this document.

provides a supplement to other sections of the ERA that discuss the uncertainty in the conclusions specific to RTE species.

6.2.1 Identification of Surrogate Species

Use of surrogate species in a screening ERA is necessary to address the broad range of species likely to be encountered on BLM-managed lands as well as to accommodate the fact that toxicity data may be restricted to a limited number of species. In this ERA, surrogates were selected to account for variation in the nature of potential herbicide exposure (e.g., direct contact, food chain) as well as to ensure that different taxa, and their behaviors, are considered. As described in Section 3.0 of the Methods document (ENSR 2004c), surrogate species were selected to represent a broad range of taxa in several trophic guilds that could potentially be impacted by herbicides on BLM-managed lands. Generally, the surrogate species that were used in the ERA are species commonly used as representative species in ERA. Many of these species are common laboratory species, or are described in USEPA (1993a, b) Exposure Factors Handbook for Wildlife. Other species were included in the California Wildlife Biology, Exposure Factor, and Toxicity Database (CA OEHHA 2003),[5] or are those recommended by USEPA OPP for tests to support pesticide registration Surrogate species were used to derive TRVs, and in exposure scenarios that involve organism size, weight, or diet, surrogate species were exposed to the herbicide in the models to represent potential impact to other species that may be present on BLM lands.

Toxicity data from surrogate species were used in the development of TRVs because few, if any, data are available that demonstrate the toxicity of chemicals to RTE species. Most reliable toxicity tests are performed under controlled conditions in a laboratory, using standardized test species and protocols; RTE species are not used in laboratory toxicity testing. In addition, field-generated data, which are very limited in number but may include anecdotal information about RTE species, are not as reliable as laboratory data because uncontrolled factors may complicate the results of the tests (e.g., secondary stressors such as unmeasured toxicants, imperfect information on rate of exposure).

As described below, inter-species extrapolation of toxicity data often produces unknown bias in risk calculations. This ERA approached the evaluation of higher trophic level species by life history (e.g., large animals vs. small animals, herbivore vs. carnivores). Then surrogate species were used to evaluate all species of similar life history potentially found on BLM-managed lands, including RTE species. This procedure was not done for plants, invertebrates, and fish, as most exposure of these species to herbicides is via direct contact (e.g., foliar deposition, dermal deposition, dermal/gill uptake) rather than ingestion of contaminated prey items. Therefore, altering the life history of these species would not result in more or less exposure.

The following subsections describe the selection of surrogate species used in two separate contexts in the ERA.

6.2.1.1 Species Selected in Development of TRVs

As presented in Appendix A of the ERA, limited numbers of species are used for toxicity testing of chemicals, including herbicides. Species are typically selected because they tolerate laboratory conditions well. The species used in laboratory tests have relatively well-known response thresholds to a variety of chemicals. Growth rates, ingestion rates, and other species-specific parameters are known; therefore, test duration and endpoints of concern (e.g., mortality, germination) have been established in protocols for many of these laboratory species. Data generated during a toxicity test, therefore, can be compared to data from other tests and relative species sensitivity can be compared. Of course, in the case of RTE species, it would be unacceptable to subject individuals to toxicity tests.

The TRVs used in the ERA were selected after reviewing available ecotoxicological literature for fluridone. Test quality was evaluated, and tests with multiple substances were not considered for the TRV. For most receptor groups, the lowest value available for an appropriate endpoint (e.g., mortality, germination) was selected as the TRV. Using

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[5] On-line http://www.oehha.org/cal_ecotox/default.htm

the most sensitive species provides a conservative level of protection for all species. The surrogate species used in the fluridone TRVs are presented in Table 6-1.

6.2.1.2 Species Selected as Surrogates in the ERA

Plants, fish, insects, and other aquatic invertebrates were evaluated on a generic level. That is, the surrogate species evaluated to create the TRVs were selected to represent all potentially exposed species. For vertebrate terrestrial animals, in addition to these surrogate species, specific species were selected to represent the populations of similar species. The species used in the ERA are presented in Table 6-2.

The surrogate terrestrial vertebrate species selected for the ERA include species from several trophic levels that represent a variety of foraging strategies. Whenever possible, the species selected are found throughout the range of land included in the EIS; all species selected are found in at least a portion of the range. The surrogate species are common species whose life histories are well documented (USEPA 1993 a, b; CA OEHHA 2003). Because species-specific data, including BW and food ingestion rates, can vary for a single species throughout its range, data from studies conducted in western states or with western populations were selected preferentially. As necessary, site-specific data can be used to estimate potential risk to species known to occur locally.

6.2.2 Surrogates Specific to Taxa of Concern

Protection levels for different species and individuals vary. Some organisms are protected on a community level; that is, slight risk to individual species may be acceptable if the community of organisms (e.g., wildflowers, terrestrial insects) is protected. Generally, community level organisms include plants and invertebrates. Other organisms are protected on a population level; that is, slight risk to individuals of a species may be acceptable if the population, as a whole, is not endangered. However, RTE species are protected as individuals; that is, risk to any single organism is considered unacceptable. This higher level of protection motivates much of the conservative approach taken in this ERA. Surrogate species were grouped by general life strategy: sessile (i.e., plants), water dwelling (i.e., fish), and mobile terrestrial vertebrates (i.e., birds, mammals, and reptiles). The approach to account for RTE species was divided along the same lines.

Plants, fish, insects, and aquatic invertebrates were assessed using TRVs developed from surrogate species. All species from these taxa (identified in Appendix C) were represented by the surrogate species presented in Table 6-1. The evaluation of terrestrial vertebrates used surrogate species to develop TRVs and to estimate potential risk using simple food chain models. Tables 6-3 and 6-4 present the listed birds and mammals found on BLM-managed lands and their appropriate surrogate species.

Very few laboratory studies have been conducted using reptiles or amphibians. Therefore, data specific to the adverse effects of a chemical on species of these taxa are often unavailable. These animals, being cold-blooded, have very different rates of metabolism than mammals or birds (i.e., they require lower rates of food consumption). Nonetheless, mammals and birds were used as the surrogate species for reptiles and adult amphibians because of the lack of data for these taxa. Fish were used as surrogates for juvenile amphibians. For each trophic level of RTE reptile or adult amphibian, a comparable mammal or bird was selected to represent the potential risks. Table 6-5 presents the 7 listed reptiles found on BLM-managed lands and the surrogate species chosen to represent them in the ERA. Table 6-6 presents the listed amphibians found on BLM-managed lands and their surrogate species.

The sensitivity of reptiles and amphibians relative to other species is generally unknown. Some information about reptilian exposures to pesticides, including herbicides, is available. The following provides a brief summary of the data (*as cited in Sparling et al. 2000*), including data for pesticides not evaluated in this ERA:

- Mountain garter snakes (*Thamnophis elegans elegans*) were exposed to the herbicide thiobencarb in the field and in the laboratory. No effects were noted in the snakes fed contaminated prey or those caged and exposed directly to treated areas.

- No adverse effects to turtles were noted in a pond treated twice with the herbicide Kuron (2,4,5-T).

- Tortoises in Greece were exposed in the field to atrazine, paraquat, Kuron, and 2,4-D. No effects were noted on the tortoises exposed to atrazine or paraquat. In areas treated with Kuron and 2,4-D, no tortoises were noted following the treatment. The authors of the study concluded it was a combination of direct toxicity (tortoises were noted with swollen eyes and nasal discharge) and loss of habitat (much of the vegetation killed during the treatment had provided important ground cover for the tortoises).

- Reptilian LD_{50} values from six organochlorine pesticides were compared to avian LD_{50} values. Of the six pesticides, five lizard LD_{50}s were higher, indicating lower sensitivity. Overlapping data were available for turtle exposure to one organochlorine pesticide; the turtle was less sensitive than the birds or lizards.

- In general, reptiles were found to be less sensitive than birds to cholinesterase inhibitors.

Unfortunately, these observations do not provide any sort of rigorous review of dose and response. On the other hand, there is little evidence that reptiles are more sensitive to pesticides than other, more commonly tested organisms.

As with reptiles, some toxicity data are available describing the effects of herbicides on amphibians. The following provides a brief summary of the data (as cited in Sparling et al. 2000):

- Leopard frog *(Rana pipiens)* tadpoles exposed to up to 0.075 mg/L atrazine showed no adverse effects.

- In a field study, it was noted that frog eggs in a pond where atrazine was sprayed nearby suffered 100% mortality.

- Common frog (*Rana temporaria)* tadpoles showed behavioral and growth effects when exposed to 0.2 to 20 mg/L cyanatryn.

- Caged common frog and common toad (*Bufo bufo)* tadpoles showed no adverse effects when exposed to 1.0 mg/L diquat or 1.0 mg/L dichlobenil.

- All leopard frog eggs exposed to 2.0 to 10 mg/L diquat or 0.5 to 2.0 mg/L paraquat hatched normally, but showed adverse developmental effects. It was noted that commercial formulations of paraquat were more acutely toxic than technical grade paraquat. Tadpoles, however, showed significant mortality when fed paraquat-treated parrot feather watermilfoil (*Myriophyllum).*

- 4-chloro-2-methylphenoaxyacetic acid (MCPA) is relatively non-toxic to the African clawed frog (*Xenopus laevis)* with an LC_{50} of 3,602 mg/L and slight growth retardation at 2,000 mg/L.

- Approximately 86% of juvenile toads died when exposed to monosodium methanearsonate (ANSAR 259® HC) at 12.5% of the recommended application rate.

- Embryo hatch success, tadpole mortality, growth, paralysis, and avoidance behavior were studied in three species of ranid frogs (*Rana* sp.) exposed to hexazinone and triclopyr. No effects were noted in hexazinone exposure up to 100 mg/L. Two species showed 100% mortality at 2.4 mg/L triclopyr; no significant mortality was observed in the third species.

No conclusions can be drawn regarding the sensitivity of amphibians to exposure to fluridone relative to the surrogate species selected for the ERA. Amphibians are particularly vulnerable to changes in their environment (chemical and physical) because they have skin with high permeability, making them at risk to dermal contact, and have complex life cycles, making them vulnerable to developmental defects during the many stages of metamorphosis. Given the very low risks to animals in the modeled exposures, it is unlikely the concentrations of fluridone predicted to occur as a result of regular herbicide usage would cause adverse effects to amphibians. Nonetheless, it should be noted that amphibians can be sensitive to pesticides, and site- and species-specific risk assessment should be carefully considered in the event that amphibian RTE species are present near a site of application.

Although the uncertainties associated with the potential risk to RTE mammals, birds, reptiles, and amphibians are valid, the vertebrate RQs generated in the ERA for fluridone are generally very low (Section 4.3). None of the RQs exceed respective LOCs. Of the four general scenarios in which vertebrate receptors were evaluated, the highest RQ was 0.38 (chronic exposure of small mammalian herbivore ingesting prey contaminated by direct spray at maximum application rate). This RQ is lower than the chronic RTE LOC of 1. Most vertebrate RQs, including fish exposure to normal applications, were lower than respective LOCs by several orders of magnitude.

6.2.3 Biological Factors Affecting Impact from Herbicide Exposure

The potential for ecological receptors to be exposed to, and affected by, herbicide is dependent upon many factors. Many of these factors are independent of the biology or life history of the receptor (e.g., timing of herbicide use, distance to receptor). These factors were explored in the ERA by simulating scenarios that vary these factors (ENSR 2004c), and these scenarios are discussed in Section 5.0 of this document. However, there are differences in life history among and between receptors that also influence the potential for exposure. Therefore, individual species have a different potential for exposure as well as response. In order to provide perspective on the assumptions made here, as well as the potential need to evaluate alternatives, receptor traits that may influence species-specific exposure and response were examined. These traits are presented and discussed in Table 6-7.

In addition to providing a review of the approach used in the ERA, the factors listed in Table 6-7 can be evaluated in order to assess whether a site- and species-specific ERA should be considered to address potential risks to a given RTE. They also provide perspective on the uncertainty associated with applying the conclusions of the ERA to a broad range of RTE species.

6.3 Review of Extrapolation Methods Used to Calculate Potential Exposure and Risk

Ecological risk assessment relies on extrapolation of observations from one system (e.g., species and toxicity endpoint) to another (see Table 6-7). While every effort has been made to anticipate bias in these extrapolations and to use them to provide an overestimate of risk, it is worth evaluating alternative approaches.

Toxicity Extrapolations in Terrestrial Systems (Fairbrother and Kaputska 1996) is an opinion paper that describes the difficulties associated with trying to quantitatively evaluate a particular species when toxicity data for that species, and for the endpoint of concern, are not available. The authors provide an overview of uncertainty factors and methods of data extrapolation used in terrestrial organism TRV development, and suggest an alternative approach to establishing inter-species TRVs. The following subsections summarize their findings for relevant methods of extrapolation.

6.3.1 Uncertainty Factors

Uncertainty factors are used often in both human health and ERA. The uncertainty factor most commonly used in ERA is 10. This value has little empirical basis, but was developed and adopted by the risk assessment community because it seemed conservative and was "simple to use."[6] Six situations in which uncertainty factors may be applied in ecotoxicology were identified: (1) accounting for intraspecific heterogeneity, (2) supporting interspecific extrapolation, (3) converting acute to chronic endpoints and vice versa, (4) estimating LOAEL from NOAEL, (5) supplementing professional judgment, and (6) extrapolating laboratory data to field conditions. No extrapolation of toxicity data among Classes (i.e., between birds, mammals, and reptiles) was discussed. The methods to extrapolate available laboratory toxicity data to suit the requirements of the TRVs in this ERA are discussed in Section 3. For this reason, extrapolation used to develop TRVs is not discussed in this section.

[6] Section 2, Fairbrother and Kaputska 1996. Page 7.

Empirical data for each of the situations discussed in the Fairbrother and Kaputska paper (as applicable) are presented in Tables 6-8 through 6-12. In each of these tables, Fairbrother and Kaputska (1996) have presented the percentage of the available data that is included within a stated factor. For example, 90% of the observed LD_{50} for bird species lie within a factor of ten (i.e., the highest LD_{50} within the central 90% of the population is 10-fold higher than the lowest value). This can be compared to the approach used in this ERA. For example, for aquatic invertebrates, a LOC was defined of 0.05. This is analogous to application of an uncertainty factor 20 to the relevant TRV. In this case, the selected TRV is not the highest or the mid-point of the available values but a value at the lower end of the available range. Thus, dividing the TRV by a factor of 20 is very likely to place it well below any observed TRV. With this perspective, the ranges (or uncertainty factors) provided by Fairbrother and Kaputska (1996) generally appear to support the approach used in the ERA (i.e., select low TRVs and consider comparison to an LOC < 1.0).

6.3.2 Allometric Scaling

Allometric scaling provides a formula based on BW that allows translation of doses from one animal species to another. In this ERA, allometric scaling was used to extrapolate the terrestrial vertebrate TRVs from the laboratory species to the surrogate species used to estimate potential risk. The Environmental Sciences Division of the Oak Ridge National Laboratory (ORNL) (Opresko et al. 1994 and Sample et al. 1996) has used allometric scaling for many years to establish benchmarks for vertebrate wildlife. The USEPA has also used allometric scaling in development of wildlife water quality criteria in the Great Lakes Water Quality Initiative (USEPA 1995) and in the development of ecological soil screening levels (USEPA 2000).

The theory behind allometric scaling is that metabolic rate is proportional to body size.[7] However, assumptions are made that toxicological processes are dependent on metabolic rate, and that toxins are equally bioavailable among species. Similar to other types of extrapolation, allometric scaling is sensitive to the species used in the toxicity test selected to develop the TRV. Given the limited amount of data, using the lowest value available for the most sensitive species is the best approach[4], although the potential remains for site-specific receptors to be more sensitive to the toxin. Further uncertainty is introduced to allometric scaling when the species-specific parameters (e.g., BW, ingestion rate) are selected. Interspecies variation of these parameters can be considerable, especially among geographic regions. Allometric scaling is not applicable between classes of organisms (i.e., bird to mammal). However, given these uncertainties, allometric scaling remains the most reliable easy-to-use means to establish TRVs for a variety terrestrial vertebrate species (Fairbrother and Kaputska 1996).

6.3.3 Recommendations

Fairbrother and Kaputska (1996) provided a critical evaluation of the existing, proposed, and potential means for intra-species toxicity value extrapolation. The paper they published describes the shortcomings of many methods of intra-specific extrapolation of toxicity data for terrestrial organisms. Using uncertainty factors or allometric scaling for extrapolation can often over- or underpredict the toxic effect to the receptor organism. Although using physiologically-based models may be a more scientifically correct way to predict toxicity, the logistics involved with applying them to an ERA on a large-scale make them impractical. In this ERA, extrapolation was performed using techniques most often employed by the scientific risk assessment community. These techniques included the use of uncertainty factors (i.e., potential use of LOC < 1.0) and allometric scaling.

6.4 Indirect Effects on Salmonids

In addition to the potential direct toxicity associated with herbicide exposure, organisms may be harmed from indirect effects, such as habitat degradation or loss of prey. Under Section 9 of the ESA of 1973, it is illegal to take an

[7] In the 1996 update to the ORNL terrestrial wildlife screening values document (Sample et al. 1996), studies by Mineau et al. (1996) using allometric scaling indicated that, for 37 pesticides studied, avian LD_{50}s varied from 1 to 1.55, with a mean of 1.148. The LD_{50} for birds is now recommended to be 1 across all species.

endangered species of fish or wildlife. "Take" is defined as "harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct" (16 USC 1532(19)). The National Marine Fisheries Service (NMFS, NOAA 1999) published a final rule clarifying the definition of "harm" as it relates to take of endangered species in the ESA. NOAA Fisheries defines "harm" as any act that injures or kills fish and wildlife. Acts may include "significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding or sheltering." To comply with the ESA, potential secondary effects to salmonids were evaluated to ensure that use of fluridone on BLM-managed lands would not cause harm to these endangered fish.

Indirect effects can generally be categorized into effects caused by biological or physical disturbance. Biological disturbance includes impacts to the food chain; physical disturbance includes impacts to habitat.[8] (Freeman and Boutin 1994).NOAA Fisheries (2002) has internal draft guidance for their Section 7 pesticide evaluations. The internal draft guidance describes the steps that should be taken in an ERA to ensure salmonids are addressed appropriately. The following subsections describe how, consistent with internal draft guidance from NOAA Fisheries, the fluridone ERA dealt with the indirect effects assessment.

6.4.1 Biological Disturbance

Potential direct effects to salmonids were evaluated in the ERA. Sensitive endpoints were selected for the RTE species RQ calculations, and worst-case scenarios were assumed. No fluridone RQs for fish in normal (i.e., not accidental) scenarios exceeded the respective RTE LOC (Section 4.3). The maximum application rate RQs for fish exposed to a spill in a pond or in a stream from accidental spray slightly exceed their respective LOCs. Indirect effects caused by disturbance to the surrounding biological system were evaluated by looking at potential damage to the food chain.

The majority of the salmonid diet consists of aquatic invertebrates. Sustaining the aquatic invertebrate population is vital to minimizing biological damage to salmonids from herbicide use. Consistent with ERA guidance (USEPA 1997, 1998), protection of non-RTE species, such as the aquatic invertebrates serving as prey to salmonids, is at the population or community level, not the individual level. Sustainability of the numbers (population) or types (community) of aquatic invertebrates is the assessment endpoint. Therefore, unless acute risks are present, it is unlikely the herbicide will cause harm to the prey base of salmonids from direct damage to the aquatic invertebrates. As discussed in Section 4.3, with the exception of accidental spills or sprays, no aquatic invertebrate chronic scenario RQs exceeded respective LOCs. The aquatic invertebrate RQ from acute exposure to maximum application rate usage in a pond slightly exceeded the LOC. However, direct or indirect effects on streams, not ponds, are of primary concern to the protection of salmonids. Overall, the results of the ERA suggest that direct impacts to the forage of salmonids is unlikely.

As primary producers and the food base of aquatic invertebrates, disturbance to the aquatic vegetation may affect the aquatic invertebrate population, thereby affecting salmonids. With the exception of the accidental spill scenario, no risks to aquatic plants are estimated in the ERA. This suggests that the potential for impacts to aquatic vegetation and potential indirect effects on salmonids from the use of the herbicide are likely to be restricted to only a few extreme scenarios such as spills.

The actual food items of many aquatic invertebrates, however, are not leafy aquatic vegetation, but detritus or benthic algae. Should aquatic vegetation be affected by an accidental herbicide exposure, the detritus in the stream may increase. Disturbance of benthic algae communities as a result of herbicide application would cause an indirect effect (i.e., reduction in biomass at the base of the food chain) on all organisms living in the waterbody, including salmonids

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[8] Physical damage to habitat may also be covered under an evaluation of critical habitat. Since all reaches of streams and rivers on BLM land may not be listed as critical habitat, a generalized approach to potential damage to any habitat was conducted. This should satisfy a general evaluation of critical habitats. Any potential for risk due to physical damage to habitat should be addressed specifically for areas deemed critical habitat.

(benthic algae are often the principal primary producers in streams). However, data for fluridone toxicity to benthic algae were not found.

Based on an evaluation of the RQs calculated for this ERA, it is unlikely RTE fish, including salmonids, would be at risk from the indirect effects this herbicide may have on the aquatic food chain. Exceptions to this include potential acute effects to aquatic life from accidental spills, an extreme and unlikely scenario considered in this ERA to add conservatism to the risk estimates. Appropriate and careful use of fluridone should preclude such an incident.

6.4.2 Physical Disturbance

The potential for indirect effects to salmonids due to physical disturbance is less easy to define that the potential for direct biological effects. Salmonids have distinct habitat requirements; any alteration to the coldwater streams in which they spawn and live until returning to the ocean as adults can be detrimental to the salmonid population. Out of the potential effects of herbicide application, it is likely the killing of instream and riparian vegetation would cause the most important physical disturbances. The potential adverse effects could include, but would not necessarily be limited to: loss of primary producers (Section 6.4.1); loss of overhead cover, which may serve as refuge from predators or shade to provide cooling to the waterbodies; and increased sedimentation due to loss of riparian vegetation.

Adverse effects caused by herbicides can be cumulative, both in terms of toxicity stress from break-down products and other chemical stressors that may be present, and in terms of the use of herbicides on lands already stressed at a larger scale. Cumulative watershed effects (CWEs) often arise in conjunction with other land use practices, such as prescribed burning.[9]. In forested areas, herbicides are generally used in areas that have been previously altered, such as cut or burned, during vegetative succession when invasive species may dominate. The de-vegetation of these previously stressed areas can delay the stabilization of the substrate, increasing the potential for erosion and resulting sedimentation in adjacent waterbodies.

No data to support the derivation of TRVs for terrestrial plants were found in the literature search. Therefore, the potential effects of fluridone accidental spray or drift onto terrestrial vegetation, including riparian cover in salmonid habitats, is not quantifiable. Having said this, land managers should consider the proximity of salmonid habitat to potential application areas. It may be productive to develop a more site- and/or species-specific ERA in order to ensure that the proposed herbicide application will not result in secondary impacts to salmonids especially associated with loss of riparian cover.

6.5 Conclusions

The fluridone ERA evaluated the potential risks to many species using many exposure scenarios. Some exposure scenarios are likely to occur, whereas others are unlikely to occur but were included to provide a level of conservatism to the ERA. Individual RTE species were not directly evaluated. Instead, surrogate species toxicity data were used to indirectly evaluate RTE species exposure. Higher trophic level receptors were also evaluated based on their life history strategies; RTE species were represented by one of several avian or mammalian species commonly used in ERA. To provide a layer of conservatism to the evaluation, lower LOCs and TRVs were used to assess the potential impacts to RTE species.

Uncertainty factors and allometric scaling were used to adjust the toxicity data on a species-specific basis when they were likely to improve applicability and/or conservatism. As discussed in Section 3.1, TRVs were developed using the best available data; uncertainty factors were applied to toxicity data consistent with recommendation of Chapman et al. (1998).

[9] The following website provides a more detailed discussion of CWEs http://www.humbolt1.com/~heyenga/Herb.Drft.8_12_99.html.

Potential secondary effects of fluridone use should be of primary concern for the protection of RTE species. Habitat disturbance and disruptions in the food chain are often the cause of population declines of species. For RTE species, habitat or food chain disruptions should be avoided to the extent practical. Some relationships among species are mutualistic, commensalistic, or otherwise symbiotic. For example, many species rely on a particular food source or habitat. Without that food or habitat species, the dependent species may be unduly stressed or extirpated. For RTE species, these obligatory habitats are often listed by USFWS as critical habitats. Critical habitats are afforded certain protection under the ESA. All listed critical habitat, as well as habitats that would likely support RTE species, should be avoided, as disturbance to the habitat may have an indirect adverse effect on RTE species.

Herbicides may reduce riparian zones or harm primary producers in the waterbodies. The results of the ERA indicate that non-target aquatic plants may be at risk from fluridone when accidents occur, such as spills. However, the effects of aquatic herbicides in water are expected to be relatively transient and stream flow is likely to reduce herbicide concentrations over time. Only very persistent pesticides would be expected to have effects beyond the year of their application. An OPP report on the impacts of a terrestrial herbicide on salmonids indicated that if a listed salmonid was not present during the year of application, there would likely be no concern (Turner 2003). Therefore, it is expected that potential adverse impacts to food and aquatic cover would not occur beyond the season of application.

Based on the results of the ERA, it is unlikely RTE species would be harmed by appropriate and responsible use of the herbicide fluridone on BLM-managed lands.

TABLE 6-1
Surrogate Species Used to Derive Fluridone TRVs

Species in Fluridone Laboratory/Toxicity Studies		Surrogate for
Honeybee	*Apis mellifera*	Pollinating insects
Mouse	*Cavia sp.*	Mammals
Rat	*Rattus norvegicus* spp.	Mammals
Dog	*Canis familiaris*	Mammals
Rabbit	*Leporidae sp*	Mammals
Mallard	*Anas platyrhynchos*	Birds
Bobwhite Quail	*Colinus virginianus*	Birds
Midge	*Chironomus tentans*	Aquatic invertebrates
Rainbow trout	*Oncorhynchus mykiss*	Fish/Salmonids
Fathead minnow	*Pimephales promelas*	Fish
American pondweed	*Potamogeton nodosus*	Non-target aquatic plants

TABLE 6-2
Surrogate Species Used in Quantitative ERA Evaluation

Species		Trophic Level/Guild	Pathway Evaluated
American robin	*Turdus migratorius*	Avian invertivore/ vermivore/ insectivore	Ingestion
Canada goose	*Branta canadensis*	Avian granivore/ herbivore	Ingestion
Deer mouse	*Peromyscus maniculatus*	Mammalian frugivore/ herbivore	Direct contact and ingestion
Mule deer	*Odocolieus hemionus*	Mammalian herbivore/ gramivore	Ingestion
Bald eagle (northern)	*Haliaeetus leucocephalus alascanus*	Avian carnivore/ piscivore	Ingestion
Coyote	*Canis latrans*	Mammalian carnivore	Ingestion

TABLE 6-3
RTE Birds and Selected Surrogates

RTE Avian Species Potentially Occurring on BLM Lands		RTE Trophic Guild	Surrogates
Marbled murrelet	*Brachyramphus marmoratus marmoratus*	Piscivore	Bald eagle
Western snowy plover	*Charadrius alexandrinus nivosus*	Insectivore/ Piscivore	American robin
Piping plover	*Charadrius melodus*	Insectivore	American robin
Mountain plover	*Charadrius montanus*	Insectivore	American robin
Southwestern willow flycatcher	*Empidonax traillii extimus*	Insectivore	American robin
Northern aplomado falcon	*Falco femoralis septentrionalis*	Carnivore	Bald eagle Coyote
Cactus ferruginous pygmy-owl	*Glaucidium brasilianum cactorum*	Carnivore	Bald eagle Coyote
Whooping crane	*Grus Americana*	Piscivore	Bald eagle
California condor	*Gymnogyps californianus*	Carnivore	Bald eagle Coyote
Bald eagle	*Haliaeetus leucocephalus*	Piscivore	Bald eagle
Brown pelican	*Pelecanus occidentalis*	Piscivore	Bald eagle
Inyo California towhee	*Pipilo crissalis eremophilus*	Omnivore [Granivore/ Insectivore]	Canada goose American robin
Coastal California gnatcatcher	*Polioptila californica californica*	Insectivore	American robin
Stellar's eider	*Polysticta stelleri*	Piscivore	Bald eagle
Yuma clapper rail	*Rallus longirostris yumanensis*	Carnivore	Bald eagle Coyote
Spectacled eider	*Somateria fischeri*	Omnivore [Insectivore/ Herbivore]	American robin Canada goose
Least tern	*Sterna antillarum*	Piscivore	Bald eagle
Northern spotted owl	*Strix occidentalis caurina*	Carnivore	Bald eagle Coyote
Mexican spotted owl	*Strix occidentalis lucida*	Carnivore	Bald eagle Coyote
Least Bell's vireo	*Vireo bellii pusillus*	Insectivore	American robin

TABLE 6-4
RTE Mammals and Selected Surrogates

RTE Mammalian Species Potentially Occurring on BLM Lands		RTE Trophic Guild	Surrogates
Sonoran pronghorn	*Antilocapra americana sonoriensis*	Herbivore	Mule deer
Pygmy rabbit	*Brachylagus idahoensis*	Herbivore	Mule deer
Marbled murrelet	*Brachyramphus marmoratus marmoratus*	Piscivore	Bald eagle
Gray wolf	*Canis lupus*	Carnivore	Coyote
Utah prairie dog	*Cynomys parvidens*	Herbivore	Deer mouse
Morro Bay kangaroo rat	*Dipodomys heermanni morroensis*	Omnivore [Herbivore/ Insectivore]	Deer mouse American robin
Giant kangaroo rat	*Dipodomys ingens*	Granivore/ Herbivore	Deer mouse
Fresno kangaroo rat	*Dipodomys nitratoides exilis*	Granivore/ Herbivore	Deer mouse
Tipton kangaroo rat	*Dipodomys nitratoides nitratoides*	Granivore/ Herbivore	Deer mouse
Stephens' kangaroo rat	*Dipodomys stephensi (incl. D. cascus)*	Granivore	Deer mouse
Southern sea otter	*Enhydra lutris nereis*	Carnivore/ Piscivore	Coyote Bald eagle
Steller sea-lion	*Eumetopias jubatus*	Carnivore/ Piscivore	Coyote Bald eagle
Sinaloan jaguarundi	*Herpailurus (=Felis) yaguarundi tolteca*	Carnivore	Coyote
Ocelot	*Leopardus (=Felis) pardalis*	Carnivore	Coyote
Lesser long-nosed bat	*Leptonycteris curosoae yerbabuenae*	Frugivore/ Nectivore	Deer mouse
Mexican long-nosed bat	*Leptonycteris nivalis*	Herbivore	Deer mouse
Canada lynx	*Lynx canadensis*	Carnivore	Coyote
Amargosa vole	*Microtus californicus scirpensis*	Herbivore	Deer mouse
Hualapai Mexican vole	*Microtus mexicanus hualpaiensis*	Herbivore	Deer mouse
Black-footed ferret	*Mustela nigripes*	Carnivore	Coyote
Riparian (=San Joaquin Valley) woodrat	*Neotoma fuscipes riparia*	Herbivore	Deer mouse
Columbian white-tailed deer	*Odocolieus virginianus leucurus*	Herbivore	Mule deer
Bighorn sheep	*Ovis canadensis*	Herbivore	Mule deer
Bighorn sheep	*Ovis canadensis californiana*	Herbivore	Mule deer
Jaguar	*Panthera onca*	Carnivore	Coyote
Woodland caribou	*Rangifer tanandus caribou*	Herbivore	Mule deer
Northern Idaho ground squirrel	*Spermophilus brunneus brunneus*	Herbivore	Deer mouse
Grizzly bear	*Ursus arctos horribilis*	Omnivore [Herbivore/ Insectivore/ Piscivore]	American robin Mule deer Bald eagle
San Joaquin kit fox	*Vulpes macrotis mutica*	Carnivore	Coyote
Preble's meadow jumping mouse	*Zapus hudsonius preblei*	Omnivore [Herbivore/ Insectivore]	Deer mouse American robin

TABLE 6-5
RTE Reptiles and Selected Surrogates

RTE Reptilian Species Potentially Occurring on BLM Lands		RTE Trophic Guild	Surrogates
New Mexican ridge-nosed rattlesnake	*Crotalus willardi obscurus*	Carnivore/ Insectivore	Coyote/Bald eagle American robin
Blunt-nosed leopard lizard	*Gambelia silus*	Carnivore/ Insectivore	Coyote/Bald eagle American robin
Desert tortoise	*Gopherus agassizii*	Herbivore	Canada goose
Giant garter snake	*Thamnophis gigas*	Carnivore/ Insectivore/ Piscivore	Coyote American robin Bald eagle
Coachella Valley fringe-toed lizard	*Uma inornata*	Insectivore	American robin

Note: Five sea turtles are also listed species in the 17 states evaluated in this ERA. However, it is unlikely any exposure to herbicide would occur to marine species.

TABLE 6-6
RTE Amphibians and Selected Surrogates

RTE Amphibious Species Potentially Occurring on BLM Lands		RTE Trophic Guild	Surrogates
California tiger salamander	*Ambystoma californiense*	Invertivore[1] Vermivore[2]	Bluegill sunfish/Rainbow trout[3] American robin[4]
Sonoran tiger salamander	*Ambystoma tigrinum stebbinsi*	Invertivore, Insectivore[1] Carnivore, Ranivore[2]	Bluegill sunfish/Rainbow trout[3] American robin[4]
Desert slender salamander	*Batrachoseps aridus*	Invertivore	American robin[4,5]
Wyoming toad	*Bufo baxteri*	Insectivore	Bluegill sunfish/Rainbow trout[3] American robin[4]
Arroyo toad (=Arroyo southwestern toad)	*Bufo californicus*	Herbivore[1] Invertivore[2]	Bluegill sunfish/Rainbow trout[3] American robin[4]
California red-legged frog	*Rana aurora draytonii*	Herbivore[1] Invertivore[2]	Bluegill sunfish/Rainbow trout[3] American robin[4]
Chiricahua leopard frog	*Rana chiricahuensis*	Herbivore[1] Invertivore[2]	Bluegill sunfish/Rainbow trout[3] American robin[4]

(1) Diet of juvenile (larval) stage.
(2) Diet of adult stage.
(3) Surrogate for juvenile stage.
(4) Surrogate for adult stage.
(5) *Bratrachoseps aridus* is a lungless salamander that has no aquatic larval stage, and is terrestrial as an adult.

TABLE 6-7

Species and Organism Traits That May Influence Herbicide Exposure and Response

Characteristic	Mode of Influence	ERA Solution
Body size	Larger organisms have more surface area potentially exposed during a direct spray exposure scenario. However, larger organisms have a smaller surface area to volume ratio, leading to a lower per body weight dose of herbicide per application event.	To evaluate potential impacts from direct spray, small organisms were selected (i.e., honeybee and deer mouse).
Habitat preference	Not all of BLM lands are subject to nuisance vegetation control.	It was assumed that all organisms evaluated in the ERA were present in habitats subject to herbicide treatment.
Duration of potential exposure /home range	Some species are migratory or present during only a fraction of year and larger species have home ranges that likely extend beyond application areas, thereby reducing exposure duration..	It was assumed that all organisms evaluated in the ERA were present within the zone of exposure full-time (i.e., home range = application area).
Trophic level	Many chemical concentrations increase in higher trophic levels.	Although the herbicides evaluated in the ERA have very low potential to bioaccumulate, BCFs were selected to estimate uptake to trophic level 3 fish (prey item for the piscivores), and several trophic levels (primary producers through top-level carnivore) were included in the ERA.
Food preference	Certain types of food or prey may be more likely to attract and retain herbicide.	It was assumed that all types of food were susceptible to high deposition and retention of herbicide.
Food ingestion rate	On a mass ingested per body weight basis, organisms with higher food ingestion rates (e.g., mammals versus reptiles) are more likely to ingest large quantities of food (therefore, herbicide).	Surrogate species were selected that consume large quantities of food, relative to body size. When ranges of ingestion rates were provided in the literature, the upper end of the values was selected for use in the ERA.
Foraging strategy	The way an organism finds and eats food can influence its potential exposure to herbicide. Organisms that consume insects or plants that are underground are less likely to be exposed via ingestion than those that consume exposed food items, such as grasses and fruits.	It was assumed all food items evaluated in the ERA were fully exposed to herbicide during spray or runoff events.
Metabolic and excretion rate	While organisms with high metabolic rates may ingest more food, they may also have the ability to excrete herbicides quickly, lowering the potential for chronic impact.	It was assumed that no herbicide was excreted readily by any organism in the ERA.
Rate of dermal uptake	Different organisms will assimilate herbicides across their skins at different rates. For example, thick scales and shells of reptiles and the fur of mammals are likely to present a barrier to uptake relative to bare skin.	It was assumed that uptake across the skin was unimpeded by scales, shells, fur, or feathers.
Sensitivity to herbicide	Species respond to chemicals differently; some species may be more sensitive to certain chemicals.	The literature was searched and the lowest values from appropriate toxicity studies were selected as TRVs. Choosing the sensitive species as surrogates for the TRV development provides protection to more species.
Mode of toxicity	Response sites to chemical exposure may not be the same among all species. For instance, the presence of aryl hydrocarbon (Ah) receptors in an organism increase its susceptibility to compounds that bind to proteins or other cellular receptors. However, not all species, even within a given taxonomic group (e.g., mammals) have Ah receptors.	Mode of toxicity was not specifically addressed in the ERA. Rather, by selecting the lowest TRV, it was assumed that all species evaluated in the ERA were also sensitive to the mode of toxicity.

TABLE 6-8
Summary of Findings: Interspecific Extrapolation Variability

Type of Data	Percentage of Data Variability Accounted for Within a Factor of:								
	2	4	10	15	20	50	100	250	300
Bird LD_{50}	--	--	90%	--	--	--	99%	100%	--
Mammal LD_{50}	--	58%	--	--	90%	--	96%	--	--
Bird and Mammal Chronic	--	--	--	--	--	94%	--	--	--
Plants	93%[(a)] 80%[(b)]	--	--	80%[(c)]	--	--	--	--	80%[(d)]

(a) Intra-genus extrapolation.
(b) Intra-family extrapolation.
(c) Intra-order extrapolation.
(d) Intra-class extrapolation.

TABLE 6-9
Summary of Findings: Intraspecific Extrapolation Variability

Type of Data	Percentage of Data Variability Accounted for Within Factor of 10	Citation from Fairbrother and Kaputska 1996
490 probit log-dose slopes	92%	Dourson and Starta 1983 *as cited in Abt Assoc., Inc. 1995*
Bird LC_{50}:LC_1	95%	Hill et al. 1975
Bobwhite quail LC_{50}:LC_1	71.5%	Shirazi et al. 1994

TABLE 6-10
Summary of Findings: Acute-to-Chronic Extrapolation Variability

Type of Data	Percentage of Data Variability Accounted for Within Factor of 10	Citation from Fairbrother and Kaputska 1996
Bird and mammal dietary toxicity NOAELs (n=174)	90%	Abt Assoc., Inc. 1995

TABLE 6-11
Summary of Findings: LOAEL-to-NOAEL Extrapolation Variability

Type of Data	Percentage of Data Variability Accounted for Within Factor of:		Citation from Fairbrother and Kaputska 1996
	6	10	
Bird and mammal LOAELs and NOAELs	80%	97%	Abt Assoc., Inc. 1995

TABLE 6-12
Summary of Findings: Laboratory to Field Extrapolations

Type of Data	Response	Citation from Fairbrother and Kaputska 1996
Plant EC_{50} Values	3 of 20 EC_{50} lab study values were 2-fold higher than field data. 3 of 20 EC_{50} values from field data were 2-fold higher than lab study data	Fletcher et al. 1990
Bobwhite quail	Shown to be more sensitive to cholinesterase-inhibitors when cold-stressed (i.e., more sensitive in the field).	Maguire and Williams 1987
Gray-tailed vole and deer mouse	Laboratory data over-predicted risk	Edge et al. 1995

7.0 UNCERTAINTY IN THE ECOLOGICAL RISK ASSESSMENT

Every time an assumption is made, some level of uncertainty is introduced into the risk assessment. A thorough description of uncertainties is a key component that serves to identify possible weaknesses in the ERA analysis, and to elucidate what impact such weaknesses might have on the final risk conclusions. This uncertainty analysis lists the uncertainties, with a discussion of what bias—if any—the uncertainty may introduce into the risk conclusions. This "bias" is represented in qualitative terms that best describe whether the uncertainty might 1) underestimate risk, 2) overestimate risk, or 3) be neutral with regard to the risk estimates, or whether it cannot be determined without additional study.

Uncertainties in the ERA process are summarized in Table 7-1. Several of the uncertainties warrant further evaluation and are discussed below. In general, the assumptions made in this risk assessment have been designed to yield a conservative evaluation of the potential risks to the environment from herbicide application.

7.1 Toxicity Data Availability

The majority of the available toxicity data was obtained from studies conducted as part of the USEPA pesticide registration process. There are a number of uncertainties related to the use of this limited data set in the risk assessment. In general, it would often be preferable to base any ecological risk analysis on reliable field studies that clearly identify and quantify the amount of potential risk from particular exposure concentrations of the chemical of concern. However, in most risk assessments it is more common to extrapolate the results obtained in the laboratory to the receptors found in the field. It should be noted, however, that laboratory studies often actually overestimate risk relative to field studies (Fairbrother and Kapustka 1996).

Only one fluridone incident report was available from the USEPAs Environmental Fate and Effects Division (EFED). Incident reports can be used to validate both exposure models and hazards to ecological receptors. This report, described in Section 2.3, listed direct contact with fluridone as the "probable" cause of tomato plant damage. No terrestrial plant toxicity data was identified in the TRV derivation process, and impacts to terrestrial plants were not assessed in the risk assessment. This incident report suggests that impacts to non-target terrestrial plants may be of concern in accidental direct spray scenario. However, the use and severity of the impact were undetermined so it is impossible to correlate the concentrations predicted by the accidental spay scenario with the incident report.

Species for which toxicity data are available may not necessarily be the most sensitive species to a particular herbicide. These species have been selected as laboratory test organisms because they are generally sensitive to stressors, yet they can be maintained under laboratory conditions. However, the selected toxicity value for a receptor was based on a thorough review of the available data by qualified toxicologists and the selection of the most appropriate sensitive surrogate species. The surrogate species used in the registration testing are not an exact match to the wildlife receptors included in the ERA. For example, the only avian data available is for two primarily herbivorous birds: the mallard duck and the bobwhite quail. However, TRVs based on these receptors were also used to evaluate risk to insectivorous and piscivorous birds. Species with alternative feeding habits or species from different taxonomic groups may be more or less sensitive to the herbicide than those species tested in the laboratory.

In general, the most sensitive available endpoint for the appropriate surrogate test species was used to derive TRVs. This is a conservative approach since there may be a wide range of data and effects for different species. This selection criterion for the TRVs has the potential to overestimate risk within the ERA. In some cases (i.e., coldwater fish), chronic data was unavailable and chronic TRVs were derived from acute toxicity data, adding an additional level of conservatism.

There is also some uncertainty in the conversion of food concentration-based toxicity values (mg herbicide per kg food) to dose-based values (mg herbicide per kg BW) for birds and mammals. Converting the concentration-based endpoint to a dose-based endpoint is dependent upon certain assumptions, specifically the test animal ingestion rate and test animal BW. Default ingestion rates for different test species were used in the conversions unless test-specific values were measured and given. The ingestion rate was assumed to be constant throughout a test. However, it is possible that a test chemical may positively or negatively affect ingestion, thus resulting in an over- or underestimation of total dose.

For the purposes of pesticide registration, tests are conducted according to specific test protocols. For example, in the case of an avian oral LD_{50} study, test guidance follows the harmonized Office of Pollution Prevention and Toxic Substances (OPPTS) protocol 850.2100, Avian Acute Oral Toxicity Test or its Toxic Substances Control Act (TSCA) or FIFRA predecessor (e.g, 40 CFR 797.2175 and OPP 71-1). In this test the bird is given a single dose, by gavage, of the chemical and the test subject is observed for a minimum of 14 days. The LD_{50} derived from this test is the true dose (mg herbicide per kg BW). However, dietary studies were selected preferentially for this ERA and historical dietary studies followed 40 CFR 797.2050, OPP 71-2, or OECD 205, the procedures for which are harmonized in OPPTS 850.2200, Avian Dietary Toxicity Test. In this test, the test organism is presented with the dosed food for 5 days, with 3 days of additional observations after the chemical-laden food is removed. The endpoint for this assay is reported as an LC_{50} representing mg herbicide per kg food. For this ERA, the concentration-based value was converted to a dose-based value following the methodology presented in the Methods Document (ENSR 2004c)[10]. Then the dose-based value was multiplied by the number of days of exposure (generally 5) to result in an LD_{50} value representing the full herbicide exposure over the course of the test.

For fluridone, no toxicity data was identified for terrestrial plant species. This is a type of testing generally required for pesticide registrations, but no information was identified in the FOIA review or other sources. This results in a data gap, and therefore no quantitative evaluation of potential risks to non-target terrestrial plants was possible in the risk assessment. As discussed above, one ecological incident was reported, which associated impacts to tomato plants with fluridone. In addition the manufacturer's user's guide for the Sonar aquatic herbicide (Eli Lilly and Company 2003), indicated that some upland terrestrial species (i.e., grasses, sedges) are considered to be "sensitive" or "intermediate" in their tolerance to the herbicide, while shoreline plants, (i.e., willow, cypress), were considered "tolerant." The Sonar labels (SePRO 2002a,b,c; SePRO 2003) warn against using treated water for irrigation purposes for seven to thirty days after treatment. Even at the low fluridone concentrations used to treat milfoil, some terrestrial plants may be sensitive to fluridone if they are watered with treated lake water. The incident report, the user's guide, and the herbicide labels indicate that fluridone may cause negative impacts to terrestrial plants (e.g., tomatoes, grasses, sedges), but that shoreline plants are more tolerant. It is these more tolerant shoreline plants that are more likely to come in contact with fluridone during normal pond applications.

As indicated in Section 3.1, the toxicity data within the ERAs are presented in the units used in the reviewed studies. Attempts were not made to adjust toxicity data to the % a.i. since it was not consistently provided in all reviewed materials. In most cases the toxicity data applies to the a.i. itself; however, some data corresponds to a specific product containing the a.i. under consideration, and potentially other ingredients (e.g., other a.i. or inert ingredients). The assumption has been made that the toxicity observed in the tests is due to the a.i. under consideration. However, it is possible that the additional ingredients in the different formulations also had an effect. The OPP's Ecotoxicity Database (a source of data for the ERAs) does not adjust the toxicity data to the % a.i. and presents the data directly from the registration study in order to capture the potential effect caused by various inerts, additives, or other a.i. in the tested product. In many cases the tested material represents the highest purity produced and higher exposure to the a.i. would not be likely.

For fluridone, the percent a.i., listed in Appendix A when available from the reviewed study, ranged from 0.48% to 99%. The lowest % a.i. used in the actual TRV derivation was 33.3% in the study used to derive the acute TRV for the honeybee. Adjusting the TRV to 100% of the a.i. (by multiplying the TRV by the % a.i. in the study) would lower

[10] Dose-based endpoint $_{(mg/kg\ BW/day)}$ = [Concentration-based endpoint $_{(mg/kg\ food)}$ x Food Ingestion Rate $_{(kg\ food/day)}$]/BW $_{(kg)}$

the bee TRV from 1,088 ug/bee to 362 ug/bee. Although this would increase the associated RQs, it would not result in any additional LOC exceedances. The remaining TRVs are based on studies with at least 95% a.i., so the RQ changes would be minimal. Several of the fish studies included in Appendix A were conducted with products containing 41 to 48% fluridone. However, to reduce the uncertainties in whether the toxicity in these studies was due to fluridone or to other components, the values selected to derive the fish TRVs were based on studies containing 89 to 99% fluridone. Selection of alternative studies and adjustment to reflect the % a.i. could result in a lower TRV[11], but there would be a level of uncertainty in this TRV due to the potential toxicity of the other components in the product.

7.2 Potential Indirect Effects on Salmonids

No actual field studies or ecological incident reports related to the effects of fluridone on salmonids were identified during the ERA. Therefore, any discussion of direct or indirect impacts to salmonids was limited to qualitative estimates of potential impacts to salmonid populations and communities. The acute fish TRV used in the risk assessment was based on laboratory studies conducted with a salmonid, the rainbow trout, reducing the uncertainties in this evaluation.

A discussion of the potential indirect impacts to salmonids is presented in Section 4.3.6, and Section 6.6 provides a discussion of RTE salmonid species. These evaluations indicated that, in the conservative accidental exposure scenarios evaluated, salmonids may be indirectly impacted by a reduction in food supply (i.e., fish and aquatic invertebrates), but not a reduction in aquatic vegetative cover.

It is anticipated that these qualitative evaluations over-estimate the potential risk to salmonids due to the conservative selection of TRVs for salmonid prey and vegetative cover, application of additional LOCs (with uncertainty/safety factors applied) to assess risk to RTE species, and the use of conservative stream characteristics in the exposure scenarios (i.e., low order stream, relatively small instantaneous volume, limited consideration of herbicide degradation or absorption in models).

7.3 Ecological Risks of Degradates, Inert Ingredients, Adjuvants, and Tank Mixtures

In a detailed herbicide risk assessment, it is preferable to estimate risks not just from the a.i. of an herbicide, but also from the cumulative risks of inert ingredients (inerts), adjuvants, surfactants, and degradates. Other herbicides may also factor into the risk estimates, as many herbicides can be tank mixed to expand the level of control and to accomplish multiple identified tasks. However, using currently available models (e.g., AgDRIFT®), it is only practical to calculate deterministic risk calculations (i.e., exposure modeling, effects assessment, and RQ calculations) for a single a.i.

In addition, information on inerts, adjuvants, and degradates is often limited by the availability of, and access to, reliable toxicity data for these constituents. The sections below present a qualitative evaluation of potential effects for risks from inert ingredients, adjuvants, and tank mixtures.

7.3.1 Degradates

The potential toxicity of degradates, also called herbicide transformation products (TPs), should be considered when selecting an herbicide. However, it is beyond the scope of this risk assessment to evaluate all of the possible degradates of the various herbicide formulations containing fluridone. Degradates may be more or less mobile and

[11] Selection of the channel catfish study conducted using 41% fluridone and adjustment of that 96 hour LC_{50} (13.2 mg/L) to reflect the % active ingredient would result in a warm water fish acute TRV of 5.4 mg/L. This value is lower than the selected value of 8.2 mg/L conducted with a product containing 98 to 99% fluridone.

more or less toxic in the environment than their source herbicides (Battaglin et al. 2003). Differences in environmental behavior (e.g., mobility) and toxicity between parent herbicides and TPs makes prediction of potential TP impacts challenging. For example, a less toxic, but more mobile bioaccumulative, or persistent TP may have the potential to have a greater adverse impact on the environment resulting from residual concentrations in the environment. A recent study indicated that 70% of TPs had either similar or reduced toxicity to fish, daphnids, and algae than the parent pesticide. However, 4.2% of the TPs were more than an order of magnitude more toxic than the parent pesticide, with a few instances of acute toxicity values below 1 mg/L (Sinclair and Boxall 2003). No evaluation of impacts to terrestrial species was conducted in this study. The lack of data on the toxicity of degradates of fluridone represents a source of uncertainty in the risk assessment.

7.3.2 Inerts

Pesticide products contain both active and inert ingredients. The terms "active ingredient" and "inert ingredient" have been defined by Federal law—the FIFRA—since 1947. An a.i. is one that prevents, destroys, repels or mitigates the effects of a pest, or is a plant regulator, defoliant, desiccant, or nitrogen stabilizer. By law, the a.i. must be identified by name on the label, together with its percentage by weight. An inert ingredient is simply any ingredient in the product that is not intended to affect a target pest. For example, isopropyl alcohol may be an a.i. and antimicrobial pesticide in some products; however, in other products, it is used as a solvent and may be considered an inert ingredient. The law does not require inert ingredients to be identified by name and percentage on the label, but the total percentage of such ingredients must be declared.

In September 1997, the USEPA issued Pesticide Regulation Notice 97-6, which encouraged manufacturers, formulators, producers, and registrants of pesticide products to voluntarily substitute the term "other ingredients" as a heading for the inert ingredients in the ingredient statement. The USEPA made this change after learning the results of a consumer survey on the use of household pesticides. Many consumers are misled by the term "inert ingredient," believing it to mean "harmless." Since neither the federal law nor the regulations define the term "inert" on the basis of toxicity, hazard or risk to humans, non-target species, or the environment, it should not be assumed that all inert ingredients are non-toxic. Whether referred to as "inerts" or "other ingredients," these components within an herbicide have the potential to be toxic.

BLM scientists received clearance from the USEPA to review CBI on inert compounds in the following herbicides under consideration in ERAs: bromacil, chlorsulfuron, diflufenzopyr, Overdrive® (a mix of dicamba and diflufenzopyr), diquat, diuron, fluridone, imazapic, sulfometuron-methyl, and tebuthiuron. The information received listed the inert ingredients, their chemical abstract number, supplier, USEPA registration number, percentage of the formulation and purpose in the formulation. This information is confidential, and is therefore not disclosed in this document. However, a review of available data for the herbicides is included in Appendix D.

The USEPA has a listing of regulated inert ingredients at http://www.epa.gov/opprd001/inerts/index.html. This listing categorizes inert ingredients into four lists. The listing of categories and the number of inert ingredients found among the ingredients listed for the herbicides are shown below:

- List 1 – Inert Ingredients of Toxicological Concern: None.
- List 2 – Potentially Toxic Inert Ingredients: None.
- List 3 – Inerts of Unknown Toxicity. 12.
- List 4 – Inerts of Minimal Toxicity. Over 50.

Nine inerts were not found on EPA's lists.

Toxicity information was also searched in the following sources:

- TOMES (a proprietary toxicological database including EPA's IRIS, the Hazardous Substance Data Bank, the Registry of Toxic Effects of Chemical Substances [RTECS]).
- EPA's ECOTOX database, which includes AQUIRE (a database containing scientific papers published on the toxic effects of chemicals to aquatic organisms).
- TOXLINE (a literature searching tool).
- Material Safety Data Sheets (MSDS) from suppliers.
- Other sources, such as the Farm Chemicals Handbook.
- Other cited literature sources.

Relatively little toxicity information was found. A few acute studies on aquatic or terrestrial species were reported. No chronic data, no cumulative effects data and almost no indirect effects data (food chain species) were found for the inerts in the herbicides.

A number of the List 4 compounds (Inerts of Minimal Toxicity) are naturally-occurring earthen materials (e.g. clay materials or simple salts) that would produce no toxicity at applied concentrations. However, some of the inerts, particularly the List 3 compounds and unlisted compounds, may have moderate to high potential toxicity to aquatic species based on MSDSs or published data.

As a tool to evaluate List 3 and unlisted inerts in the ERA, the exposure concentration of the inert compound was calculated and compared to toxicity information. As described in more detail in Appendix D, toxicity information from the above sources was used in addition to the work of Muller (1980), Lewis (1991), Dorn et al. (1997), and Wong et al. (1997) concerning aquatic toxicity of surfactants. These sources generally suggested that acute toxicity to aquatic life for surfactants and anti-foam agents ranged from 1 to 10 mg/L, and that chronic toxicity ranged as low as 0.1 mg/L.

Appendix D presents the following general observation for fluridone: low application rates for fluridone resulted in low exposure concentrations of inerts of much < 1 mg/L in all modeled cases. This indicates that inerts associated with the application of fluridone are not predicted to occur at levels that would cause acute toxicity to aquatic life. However, given the lack of specific inert toxicity data, it is not possible to state that the inerts in fluridone will not result in adverse ecological impacts. It is assumed that toxic inerts would not represent a substantial percentage of the herbicide, and that minimal impacts to the environment would result from these ingredients.

7.3.3 Adjuvants and Tank Mixtures

Evaluating the potential additional/cumulative risks from mixtures and adjuvants of pesticides is substantially more difficult than evaluating the inerts in the herbicide composition. While many herbicides are present in the natural environment along with other pesticides and toxic chemicals, the composition of such mixtures is highly site-specific, and thus nearly impossible to address at the level of the programmatic EIS.

Herbicide label information indicates whether a particular herbicide can be tank mixed with other pesticides. Adjuvants, such as surfactants, crop oil concentrates, fertilizers, etc., may also be added to the spray mixture to improve the herbicide efficacy. Without product specific toxicity data, it is impossible to quantify the potential impacts of these mixtures. In addition, a quantitative analysis could only be conducted if reliable scientific evidence allowed a determination of whether the joint action of the mixture was additive, synergistic, or antagonistic. Such evidence is not likely to exist unless the mode of action is common among the chemicals and receptors.

7.3.3.1 Adjuvants

Adjuvants generally function to enhance or prolong the activity of an a.i. For terrestrial herbicides, adjuvants aid in the absorption of the a.i. into plant tissue. Adjuvant is a broad term and includes surfactants, selected oils, anti-foaming agents, buffering compounds, drift control agents, compatibility agents, stickers, and spreaders. Adjuvants are not under the same registration guidelines as pesticides and the USEPA does not register or approve the labeling of spray adjuvants. Individual herbicide labels identify which types of adjuvants are approved for use with a particular herbicide.

In reviewing the labels of the a.i. fluridone, it is noted that there is not discussion regarding the addition of an adjuvant, indicating that the herbicide does not need to have an adjuvant added to the spray mixture in order to manage the vegetation. If an adjuvant is considered in the future, it is recommended that a compound with low toxicity and low required volumes be selected to reduce the potential for the adjuvant to influence the toxicity of the herbicide.

7.3.3.2 Tank Mixtures

In reviewing various labels of the different formulations of fluridone, the tank mixing of other aquatic herbicides is presented as an option, but the specific a.i. are not identified. However, it is not generally within BLM practice to tank mix fluridone with any other products. Therefore, additional modeling of tank mixes was not performed for fluridone.

In general it may be noted that selection of tank mixes, like adjuvants, is under the control of BLM land managers. To reduce uncertainties and potential negative impacts, it is required that land managers follow all label instructions and abide by any warnings. Labels for tank mixed products should be thoroughly reviewed and mixtures with the least potential for negative effects should be selected. This is especially relevant when a mixture is applied in a manner that may already have the potential for risk from an individual herbicide (e.g., runoff to ponds in sandy watersheds). Use of a tank mix under these conditions is likely to increase the level of uncertainty in the potential unintended risk to the environment.

7.4 Uncertainty Associated with Herbicide Exposure Concentration Models

The ERA relies on different models to predict the off-site impacts of herbicide use. These models have been developed and applied in order to develop a conservative estimate of herbicide loss from the application area to the off-site locations.

As in any screening or higher-tier ERA, a discussion of potential uncertainties from fate and exposure modeling is necessary to identify potential overestimates or underestimates of risk. In particular, the uncertainty analysis focused on which environmental characteristics (e.g., soil type, annual precipitation) exert the biggest numeric impact on model outputs. This has important implications not only for the uncertainty analysis itself, but also for the ability to apply risk calculations to different site characteristics from a risk management point of view.

7.4.1 AgDRIFT®

Off-site spray drift and resulting terrestrial deposition rates and waterbody concentrations (hypothetical pond or stream) were predicted using the computer model, AgDRIFT® Version 2.0.05 (SDTF 2002). As with any complex ERA model, a number of simplifying assumptions were made to ensure that the risk assessment results would be protective of most environmental settings encountered in the BLM land management program.

Predicted off-site spray drift and downwind deposition can be substantially altered by a number of variables intended to simulate the herbicide application process including, but not limited to: nozzle type used in the spray application of an herbicide mixture; ambient wind speed; release height (application boom height); and evaporation. Hypothetically, any variable in the model that is intended to represent some part of the physical process of spray drift and deposition can substantially alter predicted downwind drift and deposition patterns. Recognizing the lack of absolute knowledge regarding all of the scenarios likely to be encountered in the BLM land management program, these assumptions were developed to be conservative and likely result in overestimation of actual off-site spray drift and environmental impacts.

7.5 Summary of Potential Sources of Uncertainty

The analysis presented in this section has identified several potential sources of uncertainty that may introduce bias into the risk conclusions. This bias has the potential to 1) underestimate risk, 2) overestimate risk, or 3) be neutral with regard to the risk estimates, or be undetermined without additional study. In general, few of the sources of uncertainty in this ERA are likely to underestimate risk to ecological receptors. Risk is more likely to be overestimated or the impacts of the uncertainty may be neutral or impossible to predict.

The following bullets summarize the potential impacts on the risk predictions based on the analysis presented above:

- Toxicity Data Availability – Although the species for which toxicity data are available may not necessarily be the most sensitive species to a particular herbicide, the TRV selection methodology has focused on identifying conservative toxicity values that are likely to be protective of most species; the use of various LOCs contributes an additional layer of protection for species that may be more sensitive than the tested species (i.e., RTE species).

- Potential Indirect Effects on Salmonids – Only a qualitative evaluation of indirect risk to salmonids was possible since no relevant studies or incident reports were identified; it is likely that this qualitative evaluation overestimates the potential risk to salmonids due to the numerous conservative assumptions related to TRVs and exposure scenarios, and the application of additional LOCs (with uncertainty/safety factors applied) to assess risk to RTE species.

- Ecological Risks of Degradates, Inerts, Adjuvants, and Tank Mixtures – Only limited information is available regarding the toxicological effects of degradates, inerts, adjuvants, and tank mixtures; in general, it is unlikely that highly toxic degradates or inerts are present in approved herbicides. Also, selection of tank mixes and adjuvants is under the control of BLM land managers and to reduce uncertainties and potential risks products should be thoroughly reviewed and mixtures with the least potential for negative effects should be selected.

- Uncertainty Associated with Herbicide Exposure Concentration Models – Environmental characteristics (e.g., soil type, annual precipitation) will impact the three models used to predict the off-site impacts of herbicide use (i.e., AgDRIFT, GLEAMS, CALPUFF); in general, the assumptions used in the models were developed to be conservative and likely result in overestimation of actual off-site environmental impacts.

- General ERA Uncertainties – The general methodology used to conduct the ERA is more likely to overestimate risk than to underestimate risk due to the use of conservative assumptions (i.e., entire home range and diet is assumed to be impacted, aquatic waterbodies are relatively small, herbicide degradation over time is not applied in most scenarios).

TABLE 7-1
Potential Sources of Uncertainty in the ERA Process

Potential Source of Uncertainty	Direction of Effect	Justification
Physical-chemical properties of the active ingredient	Unknown	Available sources were reviewed for a variety of parameters. However, not all sources presented the same value for a parameter (e.g., water solubility) and some values were estimated.
Food chain assumed to represent those found on BLM lands	Unknown	BLM lands cover a wide variety of habitat types. A number of different exposure pathways have been included, but additional pathways may occur within management areas.
Receptors included in food chain model assumed to represent those found on BLM lands	Unknown	BLM lands cover a wide variety of habitat types. A number of different receptors have been included, but alternative receptors may occur within management areas.
Food chain model exposure parameter assumptions	Unknown	Some exposure parameters (e.g., body weight, food ingestion rates) were obtained from the literature and some were estimated. Efforts were made to select exposure parameters representative of a variety of species or feeding guilds.
Assumption that receptor species will spend 100% of time in impacted aquatic or terrestrial area (home range = application area)	Overestimate	These model exposure assumptions do not take into consideration the ecology of the wildlife receptor species. Organisms will spend varying amounts of time in different habitats, thus affecting their overall exposures. Species are not restricted to one location within the application area, may migrate freely off-site, may undergo seasonal migrations (as appropriate), and are likely to respond to habitat quality in determining foraging, resting, nesting, and nursery activities. A likely overly conservative assumption has been made that wildlife species obtain all their prey items from the application area.
Waterbody characteristics	Overestimate	The pond and stream were designed with conservative assumptions resulting in relatively small volumes. Larger waterbodies are likely to exist within application areas.
Extrapolation from test species to representative wildlife species	Unknown	Species differ with respect to absorption, metabolism, distribution, and excretion of chemicals. The magnitude and direction of the difference may vary with species. It should be noted, though, that in most cases, laboratory studies actually overestimate risk relative to field studies (Fairbrother and Kapustka 1996).
Consumption of contaminated prey	Unknown	Toxicity to prey receptors may result in sickness or mortality. Fewer prey items would be available for predators. Predators may stop foraging in areas with reduced prey populations, discriminate against, or conversely, select contaminated prey.
No evaluation of inhalation exposure pathways	Underestimate	The inhalation exposure pathways are generally considered insignificant due to the low concentration of contaminants under natural atmospheric conditions. However, under certain conditions, these exposure pathways may occur.
Assumption of 100% drift for chronic ingestion scenarios	Overestimate	It is unlikely that 100% of the application rate would be deposited on a plant or animal used as prey by another receptor. As indicated with the AgDRIFT® model (used to evaluate other herbicides in the EIS), off-site drift is only a fraction of the applied amount.
Ecological exposure concentration	Overestimate	It is unlikely any receptor would be exposed continuously to the full predicted EEC.
Oversimplification of dietary composition in food web models	Unknown	Assumptions were made that contaminated prey (e.g., vegetation, fish) were the primary prey items for wildlife. In reality, other prey items are likely consumed by these organisms.

TABLE 7-1 (Cont.)
Potential Sources of Uncertainty in the ERA Process

Potential Source of Uncertainty	Direction of Effect	Justification
Degradation or adsorption of herbicide	Overestimate	Risk estimates for direct spray and off-site drift scenarios generally do not consider degradation or adsorption. Concentrations will tend to decrease over time from degradation. Organic carbon in water or soil/sediment may bind to herbicide and reduce bioavailability.
Bioavailability of herbicides	Overestimate	Most risk estimates assume a high degree of bioavailability. Environmental factors (e.g., binding to organic carbon, weathering) may reduce bioavailablity.
Limited evaluation of dermal exposure pathways	Unknown	The dermal exposure pathway is generally considered insignificant due to natural barriers found in fur and feathers of most ecological receptors. However, under certain conditions (e.g., for amphibians), these exposure pathways may occur.
Amount of receptor's body exposed	Unknown	More or less than ½ of the honeybee or small mammal may be affected in the accidental direct spray scenarios.
Lack of toxicity information for amphibian and reptile species	Unknown	Information is not available on the toxicity of herbicides to reptiles and amphibians resulting from dietary or direct contact exposures.
Lack of toxicity information for RTE species	Unknown	Information is not available on the toxicity of herbicides to RTE species resulting from dietary or direct contact exposures. Uncertainty factors have been applied to attempt to assess risk to RTE receptors. See Section 7.2 for additional discussion of salmonids.
Safety factors applied to TRVs	Overestimate	Assumptions regarding the use of 3-fold uncertainty factors are based on precedent, rather than scientific data.
Use of lowest toxicity data to derive TRVs	Overestimate	The lowest data point observed in the laboratory may not be representative of the actual toxicity that might occur in the environment. Using the lowest reported chronic toxicity data point as a benchmark concentration is a very conservative approach, especially when there is a wide range in reported toxicity values for the relevant species. See Section 7.1 for additional discussion.
Use of NOAELs	Overestimate	Use of NOAELs may over-estimate effects since this measurement endpoint does not reflect any observed impacts. LOAELs may be orders of magnitudes above observed literature-based NOAELs, yet NOAELs were generally selected for use in the ERA.
Use of chronic exposures to estimate effects of herbicides on receptors	Overestimate	Chronic toxicity screening values assume that ecological receptors experience continuous, chronic exposure. Exposure in the environment is unlikely to be continuous for many species that may be transitory and move in and out of areas of maximum herbicide concentration.
Use of measures of effect	Overestimate	Although an attempt was made to have measures of effect reflect assessment endpoints, limited available ecotoxicological literature resulted in the selection of certain measures of effect that may overestimate assessment endpoints.
Lack of toxicity information for mammals or birds	Unknown	TRVs for certain receptors were based on a limited number of studies conducted primarily for pesticide registration. Additional studies may indicate higher or lower toxicity values. See Section 7.1 for additional discussion.
Lack of seed germination toxicity information	Unknown	TRVs were based on a limited number of studies conducted primarily for pesticide registration. A wide range of germination data was not always available. Emergence or other endpoints were also used and may be more or less sensitive to the herbicide.

TABLE 7-1 (Cont.)
Potential Sources of Uncertainty in the ERA Process

Potential Source of Uncertainty	Direction of Effect	Justification
Species used for testing in the laboratory assumed to be equally sensitive to herbicide as those found within application areas.	Unknown	Laboratory toxicity tests are normally conducted with species that are highly sensitive to contaminants in the media of exposure. Guidance manuals from regulatory agencies contain lists of the organisms that they consider to be sensitive enough to be protective of naturally occurring organisms. However, reaction of all species to herbicides is not known, and species found within application areas may be more or less sensitive than those used in the laboratory toxicity testing. See Section 7.1 for additional discussion.
Risk evaluated for individual receptors only	Overestimate	Effects on individual organisms may occur with little population or community level effects. However, as the number of affected individuals increases, the likelihood of population-level effects increases.
Lack of predictive capability	Unknown	The RQ approach provides a conservative estimate of risk based on a "snapshot" of conditions; this approach has no predictive capability.
Unidentified stressors	Unknown	It is possible that physical stressors other than those measured may affect ecological communities.
Effect of decreased prey item populations on predatory receptors	Unknown	Adverse population effects to prey items may reduce the foraging population for predatory receptors, but may not necessarily adversely impact the population of predatory species.
Multiple conservative assumptions	Overestimate	Cumulative impact of multiple conservative assumptions predicts high risk to ecological receptors.
Impact of the other ingredients (e.g., inerts, adjuvants) in the application of the herbicide	Unknown	Only the active ingredient has been investigated in the ERA. Inerts, adjuvants, and tank mixtures may increase or decrease the impacts of the active ingredient. These uncertainties are discussed further in Section 7.3.

8.0 SUMMARY

Based on the ERA conducted for fluridone, there is the potential for risk to selected ecological receptors from exposure to herbicides under specific conditions on BLM-managed lands. Table 8-1 summarizes the relative magnitude of risk predicted for ecological receptors for each route of exposure. This was accomplished by comparing the RQs against the most conservative LOC, and ranking the results for each receptor-exposure route combination from 'no potential' to 'high potential' for risk. As expected, accidental exposure scenarios (i.e., direct spray and accidental spills) may result in risk for non-target species (i.e., fish, aquatic invertebrates).

The following bullets summarize the risk assessment findings for fluridone under these conditions:

- Direct Spray – No acute risks were predicted for terrestrial wildlife (i.e., insects, birds, or mammals). Chronic risk was only predicted for one receptor scenario, the small mammalian herbivore at the maximum application rate. All other terrestrial animal exposure scenarios had RQs below the associated LOC. Risks to terrestrial plants could not be evaluated as a result of a lack of toxicity information; however, one ecological incident report suggests the potential for risk to terrestrial plants. No risks to non-target aquatic plants are predicted when waterbodies are accidentally (streams) or intentionally (ponds) sprayed, but risks to fish or aquatic invertebrates may occur when waterbodies are accidentally or intentionally sprayed.

- Off-Site Drift to Non-Target Terrestrial Plants – Risks to terrestrial plants could not be evaluated because of a lack of toxicity information; however, product literature and one ecological incident report suggest the potential for risk.

- Accidental Spill to Pond – Risk to fish, aquatic invertebrates, and non-target aquatic plants may occur when herbicides are spilled directly into the pond.

Based on the results of the ERA, it is unlikely that RTE species would be harmed by appropriate use of the herbicide fluridone on BLM-managed lands.

8.1 Recommendations

The following recommendations are designed to reduce potential unintended impacts to the environment from the application of fluridone:

- Select adjuvants carefully (none are currently ingredients in fluridone-containing Sonar products) since these have the potential to increase the level of toxicity above that predicted for the a.i. alone. This is especially important for application scenarios that already predict potential risk from the a.i. itself.

- Review, understand, and conform to "Environmental Hazards" section on herbicide label. This section warns of known pesticide risks to wildlife receptors or to the environment and provides practical ways to avoid harm to organisms or the environment.

- Avoid accidental direct spray on the stream to reduce the most significant potential impacts.

- Use the typical application rate in the pond, rather than the maximum application rate, to reduce risk to fish and aquatic invertebrates.

- Because the effects of normal herbicide application on terrestrial plants are uncertain, limit fluridone use in areas where RTE plants are near application areas. Avoid accidental direct spray and off-site drift to terrestrial plants to reduce potential impacts observed in a previous ecological incident report (Section 2.3).

- Observe buffer areas of at least 100 ft from terrestrial habitats for plane and helicopter application of fluridone if potential impacts to terrestrial RTE species are of concern.

- Limit fluridone application in wind, and monitor effects on adjacent terrestrial vegetation.

The results from this ERA assist the evaluation of proposed alternatives in the EIS and contribute to the development of a BA, specifically addressing the potential impacts to proposed and listed RTE species on western BLM treatment lands. Furthermore, this ERA will inform BLM field offices on the proper application of fluridone to ensure that impacts to plants and animals and their habitat are minimized to the extent practical.

TABLE 8-1
Typical Risk Levels Resulting from Fluridone Application

Exposure Category / Receptor Group	Direct Spray/Spill		Off-Site Drift		Surface Runoff		Wind Erosion	
	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate	Typical Application Rate	Maximum Application Rate
Terrestrial Animals	0 [16: 16]	0 [15: 16]	NE	NE	NA	NA	NA	NA
Terrestrial Plants (Typical Species)	NE	NE	NE	NE	NA	NA	NA	NA
Terrestrial Plants (RTE Species)	NE	NE	NE	NE	NA	NA	NA	NA
Fish In The Pond	0 [2: 2]	M [2: 4]	NA	NA	NA	NA	NA	NA
Fish In The Stream	0 [2: 2]	L [2: 2]	NA	NA	NA	NA	NA	NA
Aquatic Invertebrates In The Pond	0 [2: 2]	H [1: 4]	NA	NA	NA	NA	NA	NA
Aquatic Invertebrates In The Stream	L [1: 2]	M [1: 2]	NA	NA	NA	NA	NA	NA
Aquatic Plants In The Pond	0 [2: 2]	L [2: 4]	NA	NA	NA	NA	NA	NA
Aquatic Plants In The Stream	0 [2: 2]	0 [2: 2]	NA	NA	NA	NA	NA	NA
Piscivorous Bird	0 [1:1]	0 [1:1]	NA	NA	NA	NA	NA	NA

Risk Levels:
0 = No Potential for Risk (majority of RQs < most conservative LOC).
L = Low Potential for Risk (majority of RQs 1-10 times the most conservative LOC).
M = Moderate Potential for Risk (majority of RQs 10-100 times the most conservative LOC).
H = High Potential for Risk (majority of RQs >100 times the most conservative LOC).
The reported Risk Level is based on the risk level of the majority of the RQs for each exposure scenario within each of the above receptor groups and exposure categories (i.e., direct spray/spill, off-site drift, surface runoff, wind erosion). As a result, risk may be higher than the reported risk category for some scenarios within each category. The reader should consult the risk tables in Section 4 to determine the specific scenarios that result in the displayed level of risk for a given receptor group.
Number in brackets represents Number of RQs in the Indicated Risk Level: Number of Scenarios Evaluated.
NA = Not applicable. No RQs calculated for this scenario.
In cases of a tie, the more conservative (higher) risk level was selected.

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Technical Perspectives on Use of Sonar Pellet Formulations and Potential Risks to Threatened Mussels

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Mark A. Heilman, Ph.D. – Senior Aquatic Technology Leader

For thirty years, Sonar® (a.i. fluridone) has been widely used in aquatic systems, including drinking water reservoirs, many of which support state and/or federally-listed fishes and invertebrates. It has been used to selectively manage some of the most difficult to control invasive exotic plant species. Sonar has been carefully evaluated in formal environmental assessments and used in sensitive systems to restore aquatic habitat. Numerous past studies have evaluated the effect of Sonar on all stages of fishes and invertebrates, including zooplankton and benthic species (examples: Kamarianos *et al.*, 1989; Hamelink *et al.*, 1986; Parka *et al.*, 1978). Based on historical and recently available data summarized and discussed in sections below, applications of controlled-release pellet formulations of Sonar for partial-site treatments of invasive aquatic weeds are not expected to cause adverse effects to sensitive wildlife populations, particularly threatened mussel species. As supporting evidence of this conclusion, assessment outcomes and experiences from several sources are highlighted here.

In terms of aquatic invertebrate populations, sensitivity to Sonar has been carefully evaluated for a range of toxicity endpoints. The average 8-hour or 96-hour LC50 or EC50 for a variety of aquatic invertebrates (including amphipods, midges, daphnids, blue crabs, eastern oysters and pink shrimp) ranged from 3.7 to 4.3 ppm (Hamelink *et al.*, 1986). LC50 values for a variety of microscopic crustaceans including *Diaptomus*, sp., *Eucyclops* sp, *Alonella* sp., and *Cypria* sp., ranged from 8.0 - 13.0 ppm (Naqvi and Hawkins, 1989 as cited in McLaren/Hart, 1995). Daphnids, amphipods and midge larvae exposed chronically to Sonar concentrations of 0.2, 0.6 and 0.6 ppm showed no treatment related significant effects. The 21-day reproduction NOAEL for *D. magna* is 0.2 mg/L and the chronic NOAELs for *Gammarus pseudolimnaeus* (60-day growth endpoint) and *Chironomus plumosus* (30-day emergence endpoint) is 0.6 mg/L using a technical grade Sonar (Hamelink et al. 1986). This exposure scenario should be considered most extreme since the NOAEL was derived using technical grade active ingredient. The most conservative NOAEL for the most sensitive species from these assessments is over 50 times greater than the highest expected environmental concentration of Sonar in typical low-rate applications.

Several current national projects involving Sonar have focused on potential risk to federally or state-threatened molluscs. The laboratory of Dr. Greg Cope at North Carolina State University was recently engaged to directly assess select listed molluscs and their sensitivity to Sonar. Although tested freshwater molluscs were found to be more sensitive than aquatic species previously evaluated, Sonar was not toxic to either *Lampsilis siliquoidea* glochidia (48-h acute) or adult *L. fullerkati* adults (28-d static) up to 2X above maximum label rate with EC50 values of 511 ppb or higher (Archambault et al. 2015). The Archambault publication details the exposure scenarios and the discussion is in context of potential risks to endangered mussels in Lake Waccamaw (more below). That discussion is directly relevant to other aquatic weed management scenarios involving threatened mussel species. (see Appendix for full Archambault paper).

As further supporting evidence for minimal risk of Sonar use, pellet formulations have been the primary management tool of the State of California's Egeria Densa Control Program in the Sacramento-San Joaquin River Delta. The California Delta is home to numerous threatened and sensitive species, including the delta smelt *(Hypomesus transpacificus),* giant garter snake *(Thamnophis gigas),* salmonids *(Oncorhynchus spp.*), and the North American green sturgeon *(Acipenser medirostris).* As a routine re-evaluation process as part of Endangered Species Act Section 7 Consultation for use in the Delta, Sonar toxicity and effects to non-target organisms were thoroughly assessed (USDA-CBDW 2012: http://dbw.parks.ca.gov/PDF/Reports/EDCP-Biological_Assessment-131231.pdf pgs. 6-30 to 6-39 are most relevant). In support of the conclusions of the most recently assessment, no non-target impacts to fish and wildlife have been documented in 15 years of Sonar pellet use in the sensitive ecological conditions of the Delta. In addition, the California Department of Food and Agriculture (CDFA) has used Sonar pellets in many sites for multiple consecutive years over the nearly 40 years of its efforts to eradicate hydrilla from infested sites in the state. Again, this successful program has reported no adverse effects to non-target fauna. CDFA has done monitoring of sediment porewater concentrations using 'peepers' in the past in 43,000-acre Clear Lake, which has had spot treatments with Sonar pellets typically with multiple split applications totaling 120 ppb (38 lbs 5% pellets per acre in 6 feet of water). In 2009, such monitoring did show that porewater concentrations during the active treatment were elevated above levels measured in the water but averaged only 6.1 ppb with a maximum concentration of 43 ppb (Table 1). These values are well below the 150 ppb maximum label rate for in-water concentrations of Sonar and an order of magnitude or greater below toxicity values recently developed for multiple freshwater molluscs.

TABLE 1: CDFA - Clear Lake - FasTEST/Peepers/Soil OM
Collection Date: August 4, 2009 (12 days post install)

	FasTest		Peeper - Upper Cell (above sediment)		Peeper - Lower Cell (below sediment)		
Sample Site	**1 ft. from surface**	**1 m from bottom**	**Peeper 1**	**Peeper 2**	**Peeper 1**	**Peeper 2**	**% OM**
26-1	<1.0	<1.0	1.4	<1.0	19.6	4.9	2.27
26-2	<1.0	<1.0	4.1	<1.0	3.7	11.5	3.06
26-3	<1.0	<1.0	5.6	6.6	9.7	7.8	1.21
38-1	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	0.82
38-2	<1.0	<1.0	<1.0	<1.0	3.8	2.7	0.69
38-3	<1.0	<1.0	<1.0	<1.0	11.4	3.8	0.71
56-1	<1.0	<1.0	3.5	<1.0	5.2	2.9	13.10
56-2	<1.0	<1.0	1.6	1.2	1.2	<1.0	14.20
56-3	<1.0	<1.0	2.0	1.9	<1.0	1.6	16.20
69-1	<1.0	<1.0	<1.0	<1.0	4.0	**NA**	0.87
69-2	<1.0	<1.0	<1.0	1.1	2.3	2.0	1.43
69-3	<1.0	<1.0	1.0	<1.0	<1.0	42.7	0.83
77-1	<1.0	<1.0	4.4	**NA**	23.2	**NA**	1.32
77-2	<1.0	<1.0	3.3	2.0	3.7	3.6	1.24
77-3	<1.0	<1.0	2.7	1.8	10.0	11.2	0.70
Soda 1	2.3	2.8	10.2	**NA**	22.0	**NA**	6.94
Soda 2	<1.0	3.8	3.8	3.8	1.5	4.8	11.60
Soda 3	2.7	1.3	11.6	1.9	2.1	1.3	10.50
Mean*	0.72	0.86	3.21	1.52	6.94	6.79	4.87

* Assumes concentration values <1.0 are 0.5 ppb.
NA = No sample, peeper lost or insuffiecient soil

Sonar pellet formulations have been used annually starting in 2013 on Lake Waccamaw, North Carolina to eradicate hydrilla from this 'Outstanding Resource Water' that is home to many listed species of concern. NC state resource agencies formally evaluated risks of controlling hydrilla in Lake Waccamaw with Sonar pellets in 2013 and determined that the proposed Sonar application did not pose a threat to the lake's sensitive species (assessment document in Appendix). Since 2013, hydrilla control achieved by the proposed application is preserving this important and unique Carolina Bay habitat. Approximately 11% (960 infested acres) of the 8,900 acres has been treated with a total of 40 ppb of Sonar pellets with an areal rate of 6 – 16 lbs of 5% pellets per acre. Sonar concentrations over 4 years of management have been monitored every 2 – 4 weeks and have ranged between 1 – 3 ppb for up to 4+ month periods for the hydrilla eradication effort (representative results from 2014 are in Figure 1 below). The NC Wildlife Resource Commission has been conducting regular surveys of sensitive fish and mussel species in Waccamaw since 2009. Mussel survey work has been biennial and occurred in late July or early August of survey years. The last report was generated following 2015 sampling that occurred 2.5 months into the 2015 Sonar pellet treatment of hydrilla—the third of the consecutive Sonar cycles. The general conclusion of the report was that mussel populations have been stable except for an uptick in 2013 (1st cycle of Sonar pellet use) The full 2015 WRC report is included with this document (see Appendix) but one summary figure is included below highlighting the generally stable mussel populations in the lake (Figure 2). Qualitatively, the lack of impact to mussel populations in Lake Waccamaw is apparent in rake surveys conducted to locate hydrilla during treatment. Any length of rake drag along the bottom pulls up large quantities of mussels, typically the dominant Waccamaw Spike (*Elliptio waccamawensis*) (Figure 3).

In conclusion, the multiple decade history of selective aquatic weed control with Sonar and recent data from laboratory studies and field use support that Sonar—more specifically pelleted formulations—pose minimal risk to threatened mussel species when used for the management of invasive aquatic plants. Aquatic habitats severely impacted by invasive aquatic plant growth (Figure 4 – Lake Winnecunnet, Massachusetts photos from summer2016) have detrimental water quality fluctuations (large DO and pH swings and often low DO below dense surface canopies) and alteration of primary productivity (planktonic growth decreases at high macrophyte levels). It can be argued that these 'biological risks' pose much greater threat to threatened, filter-feeding mussels than pelleted Sonar. Invasive aquatic plant management with Sonar pellets provides a gradual, selective control of invasive aquatic plant biomass that minimizes water quality fluctuations. Sonar pellet formulations have been used extensively at a national level to restore aquatic habitat impacted by invasive weeds to the benefit of overall lake/reservoir ecological health, including mussels and other native organisms.

Lake Waccamaw FasTEST Results 2014

	ppb measured*								
Site ID	6/1/2014	6/12/2014	6/25/2014	7/11/2014	7/24/2014	8/13/2014	8/27/2014	9/10/2014	10/16/2014
LW14-1	1.7	0.5	0.5	1.7	1.2	1.3	1.5	1.1	0.5
LW14-2	1.1	1.7	1.2	2.8	1.3	1	1.2	1.5	0.5
LW14-3	1.7	1.4	3.4	2	1.3	1.2	1.5	1.2	0.5
LW14-4	1.4	1.5	0.5	3	1.8	1.1	*ls*	1.5	0.5
LW14-5	1.1	1.4	1.2	2.4	1.9	0.5	1.1	1.4	0.5
LW14-6	1.2	1.7	0.5	1.8	1.3	0.5	1.6	1.3	0.5
LW14-7	1.8	1.7	1.2	2.8	1.4	0.5	0.5	1.3	0.5
LW14-8	1	1.4	1.1	2.2	1.4	1	1.2	1.6	1.3
LW14-9	1.1	1.1	1	2.4	1.4	0.5	0.5	1.2	0.5
LW14-10	0.5	1.6	0.5	1.3	1.1	0.5	0.5	1.4	0.5
1 - 8 avg	1.4	1.4	1.2	2.3	1.5	0.9	1.2	1.4	0.6

*<1 noted as 0.5 ppb *ls* = lost sample

Treatment Dates: May 14, July 1, August 6

Figure 1. FasTEST Sonar herbicide measurements for 2014 treatment of Lake Waccamaw NC.

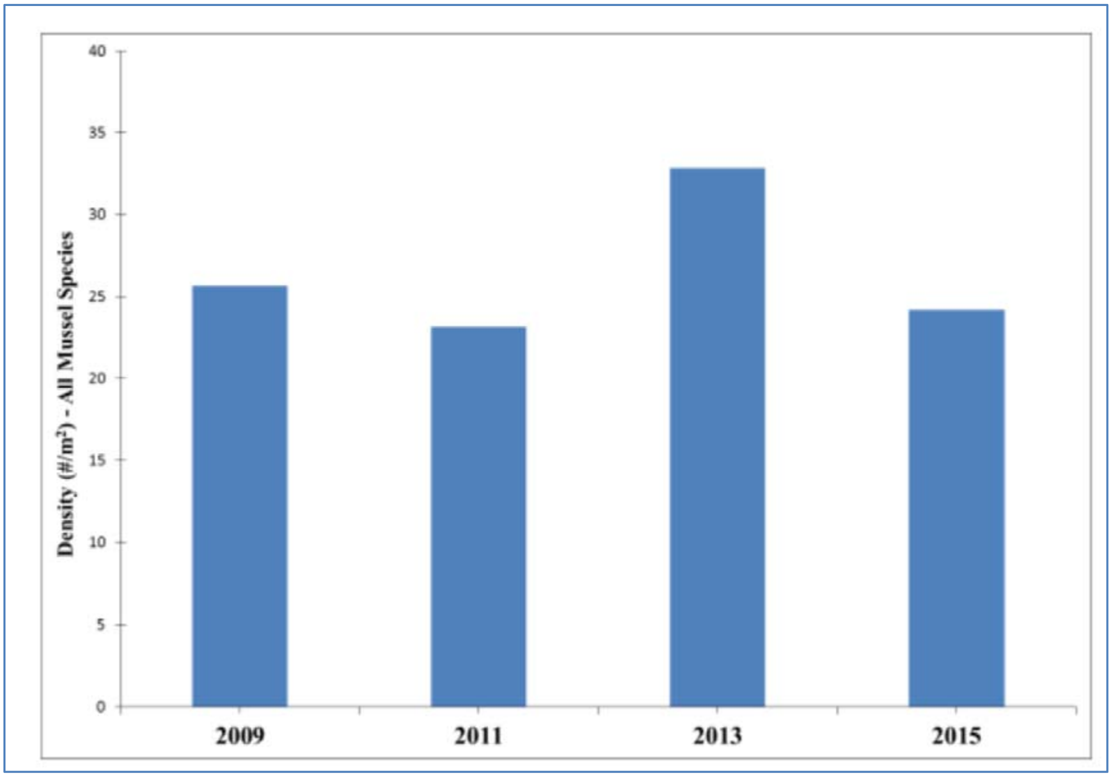

Figure 2. Annual mean density (# per square meter) of all species of mussels at monitoring sites in Lake Waccamaw from 2009 – 2015 (this is Figure 3 in 2015 WRC report in Appendix).

Figure 3. Photos of healthy Lake Waccamaw mussels incidentally pulled up by rake pulls of bottom for hydrilla detection. Photo taken July 8, 2016.

Figure 4. Photographs showing severe infestation of cabomba and variable watermilfoil in Lake Winnecunnet, Massachusetts. Pictures were taken on August 29, 2016.

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(not all used in this document but all relevant for ecological assessment of Sonar use)

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Appendix

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Sensitivity of freshwater molluscs to hydrilla-targeting herbicides: providing context for invasive aquatic weed control in diverse ecosystems

Jennifer M. Archambault[a]*, Christine M. Bergeron[a], W. Gregory Cope[a], Robert J. Richardson[b], Mark A. Heilman[c], J. Edward Corey III[d], Michael D. Netherland[e] and Ryan J. Heise[f]

[a]*Department of Applied Ecology, North Carolina State University, Raleigh, NC, USA;* [b]*Department of Crop Science, North Carolina State University, Raleigh, NC, USA;* [c]*SePRO Corporation, Carmel, NC, USA;* [d]*North Carolina Division of Parks and Recreation, Raleigh, NC, USA;* [e]*U.S. Army ERDC, Gainesville, FL, USA;* [f]*North Carolina Wildlife Resources Commission, Raleigh, NC, USA*

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Hydrilla (*Hydrilla verticillata*) is an invasive aquatic weed that has spread rapidly throughout the USA, especially in the southeast. A common control method is the application of aquatic herbicides, such as fluridone and endothall. However, there is limited documentation on the effects of herbicides commonly used to control hydrilla and other aquatic weeds on many non-target freshwater species and no published information exists on the toxicity of these herbicides to freshwater molluscs. We exposed juveniles (96 h) and glochidia (48 h) of the unionid mussel *Lampsilis siliquoidea* and adults (28 d) of *Lampsilis fullerkati* to a formulation of fluridone (Sonar – PR®) in laboratory toxicity tests. The early life stages of *L. siliquoidea* were also exposed to a formulation of the dipotassium salt of endothall (Aquathol – K®) in separate tests. Juveniles of the freshwater gastropod snail, *Somatogyrus viriginicus* (Lithoglyphidae), were exposed (96 h) to the Sonar – Genesis® fluridone formulation. Endpoints were survival (all species and life stages) as well as siphoning behavior and foot protrusion (adult mussels). Median lethal fluridone concentrations (LC50s) were 865 μg/L (95% CI, 729–1,026 μg/L) for glochidia (24 h), 511 μg/L (309–843 μg/L) for juvenile *L. siliquoidea* (96 h), and 500 μg/L (452–553 μg/L) for juvenile *S. viriginicus* (96 h). No mortality occurred in the 28-d exposure of adult *L. fullerkati* and we found no statistically significant effect of fluridone concentration on foot protrusion ($p = 0.06$) or siphoning behavior ($p = 0.08$). The 24-h LC50 for glochidia exposed to the dipotassium salt of endothall was 31.2 mg/L (30.3–32.2 mg/L) and the 96-h LC50 for juvenile mussels was 34.4 mg/L (29.3–40.5 mg/L). Freshwater molluscs were more sensitive to fluridone and endothall than most other species previously tested. Fluridone and endothall concentrations typically recommended for hydrilla treatment (5–15 μg/L and 1–5 mg/L, respectively) were not acutely toxic to the molluscs we tested and a 28-d exposure to fluridone was not lethal to adult mussels even at the highest concentration (300 μg/L), indicating minimal risk of short-term exposure effects.

Keywords: fluridone (Sonar); endothall (Aquathol); unionid mussels; snails; LC50; toxicity; invasive species

*Corresponding author. Email: jmarcham@ncsu.edu

Introduction

Freshwater systems are subject to many stressors, including point and non-point source pollution, extreme climatic events, habitat modification (e.g., dams), and invasive species (commonly anthropogenically introduced). Hydrilla (*Hydrilla verticillata*, Hydrocharitaceae) is a non-native aquatic invasive weed that was introduced into the United States in Florida in the early 1950s and has spread rapidly throughout the country, especially in the southeast (Gordon & Thomas 1997). Included on the Federal Noxious Weed List (USDA APHIS 2012), hydrilla can form vast monocultures, shade out native vegetation (FWC 2013), alter water quality parameters including dissolved oxygen (Pesacreta 1988), and can serve as a vector for a neurotoxic cyanobacteria that has been linked to avian vacuolar myelinopathy in several water birds and their predators (e.g., bald eagle *Haliaeetus leucocephalus* and great horned owl *Bubo virginianus*; Wiley et al. 2008; Williams et al. 2009). Hydrilla produces numerous vegetative propagules (e.g., tubers, turions, and shoot fragments), and is frequently dispersed by humans via boat motors, trailers, and angling gear. Given the longevity of tubers in bottom sediments, eradication and/or long-term maintenance control is difficult (Langeland 1996). The most common control methods include application of aquatic herbicides, introduction of non-native grass carp (*Ctenopharyngodon idella*), and mechanical removal (Langeland 1996). Fluridone (Sonar®), a carotenoid synthesis inhibitor herbicide, is among the most commonly used aquatic herbicides for hydrilla management, and is typically prescribed for one to four months depending on the management objective and plant maturity. The dipotassium salt of endothall (Aquathol®) is also among the most commonly used aquatic herbicides for control of hydrilla and is typically prescribed two to three times during the growing season, each for a period of days. The impetus for this study was the recent introduction and persistence of hydrilla in two North Carolina, USA, ecosystems (Lake Waccamaw and the Eno River) with high biodiversity, high rates of endemism, and the presence of threatened and endangered species (Stager & Cahoon 1987; Smith et al. 2002; NCWRC 2005; LeGrand et al. 2013; NatureServe 2013). Here, the targeted use of herbicides has been recommended as the most effective hydrilla control method that is least likely to negatively affect native vegetation. However, increased information is needed on the potential effects of these herbicides on other non-target organisms.

Lake Waccamaw is a unique Carolina Bay Lake located in the southeastern coastal plain of North Carolina, USA, because it has a neutral pH, unlike other bay lakes and blackwater systems, which enable it to support high biodiversity (Stager & Cahoon 1987). It has been called a ‘notable center of endemism in the southeast’ (Smith et al. 2002), supporting several endemic and other rare species, including two endemic unionid mussels (state-listed threatened Waccamaw fatmucket *Lampsilis fullerkati* and state-listed endangered Waccamaw spike *Elliptio waccamawensis*) and two endemic freshwater snails (Waccamaw snail *Amnicola sp. 1* and Waccamaw siltsnail *Cincinnatia sp. 1*; NCWRC 2005; LeGrand et al. 2013). The Eno River is located in the Piedmont region of North Carolina (USA), and supports a variety of rare species, including the Carolina madtom (*Noturus furiosus*, state-listed threatened), one state-threatened (*Lampsilis radiata*) and three state-endangered (*Fusconaia masoni*, *Lampsilis cariosa*, *Lasmigona subviridis*) freshwater mussels, and the only confirmed population of panhandle pebblesnail (*Somatogyrus viriginicus*) in the state (LeGrand et al. 2013).

Though toxicity data exist for some freshwater invertebrates and fishes (Crosby & Tucker 1966; Hamelink et al. 1986; Paul et al. 1994; Yi et al. 2011), to our knowledge, no information has been published on the toxicity of fluridone or endothall to freshwater molluscs. Understanding the potential risks to this non-target faunal group is especially important because both freshwater mussels and snails are simultaneously highly imperilled and critically important to the functional ecology of freshwater systems (Lydeard et al. 2004; Downing et al. 2010; Allen et al. 2012; Johnson et al. 2013). The southeastern USA, where hydrilla is most prevalent, has the highest unionid mussel biodiversity and endemism compared to any other region on the planet and >71% of North America's unionid species are endangered, threatened, or of special concern (Williams et al. 1993). Similarly, of the 703 freshwater gastropod species in USA and Canada, 278 (40%) are federally listed as endangered and >74% are considered imperilled (Johnson et al. 2013). Moreover, non-pulmonate snails and the early life stages of freshwater mussels are among the most sensitive aquatic organisms to several contaminants (e.g., atrazine, carbaryl (Conners & Black 2004); copper, ammonia (Besser et al. 2009)), and glyphosate-based chemicals which are among the most widely used herbicides (Bringolf et al. 2007). Potential risks of specific aquatic herbicides to freshwater molluscs should be assessed and balanced appropriately against the significant biological threat posed by invasive aquatic weeds like hydrilla. Further endangerment to these organisms may push some species to extinction and reduce common species to rare status.

Fluridone (market formulations tested: granular Sonar – PR® and liquid Sonar – Genesis®) and the dipotassium salt of endothall (hereafter, simply 'endothall'; market formulation Aquathol – K®) applications are commonly prescribed for management of hydrilla and both were considered for management of hydrilla in Lake Waccamaw (NC DENR 2013) and the Eno River. Unlike many aquatic systems in the southeastern USA that hydrilla has invaded and which have relatively low biodiversity (e.g., reservoirs, canals, and ponds), the two aforementioned ecologically unique systems in North Carolina, as well as others requiring similar conservation management, dictate a more thorough assessment of potential hazards to non-target biota from herbicide treatment. Therefore, we chose species of direct relevance to these systems for toxicity testing in this study. The purpose of this study was to determine the sensitivity of freshwater mussels and snails to herbicides commonly used in control and management of hydrilla and other aquatic weeds and to consider those results in the context of typically proposed hydrilla treatments (e.g., Lake Waccamaw) and potential future treatment of other sensitive ecosystems (e.g., Eno River).

Methods

Test organisms

Freshwater mussels are especially important non-target organisms for toxicity testing. They have been demonstrated as particularly susceptible to toxicants and other environmental stressors, in part because their larval life stage, glochidia, is an obligate parasite that requires encystment on a host fish to transform into the juvenile life stage (Cope et al. 2008). Therefore, juveniles and glochidia of the unionid mussel *Lampsilis siliquoidea* (fatmucket) were used in fluridone (Sonar – PR®) and endothall acute toxicity tests; *L. siliquoidea* is routinely used in toxicity testing due to its wide availability and ease of laboratory culture. *Lampsilis siliquoidea* is a congener of the Lake Waccamaw endemic and state-listed as threatened Waccamaw fatmucket (*Lampsilis fullerkati*). *Lampsilis siliquoidea* were supplied by the mussel culture laboratory at Missouri State University (Springfield, Missouri, USA). All glochidia were harvested from females <24 h before initiation of each test. All juveniles were propagated via host-fish infection, using standard propagation and culture methods (Barnhart 2006), and ranged in age from 1 to 3 d, with an average shell length of 0.25 mm (± 0.14 mm, SD).

Adult *L. fullerkati* mussels were used in a 28-d chronic experiment. They were 33 months old at the time of the experiment, with an average shell length of 46.6 mm (± 3.3 mm; range 37.5−53.9 mm) and mean weight of 9.9 g (± 2.0 g; range 6.3−14.8 g). *Lampsilis fullerkati* were propagated at the Aquatic Epidemiology Conservation Laboratory, North Carolina State University College of Veterinary Medicine (Raleigh, North Carolina, USA), using a standard *in vitro* propagation protocol (Owen 2009).

The freshwater snail *Somatogyrus virginicus* was used in acute toxicity tests with the Sonar – Genesis® fluridone formulation; *S. viriginicus* (Lithoglyphidae) is a rare, non-pulmonate snail with patchy distribution in Atlantic Slope streams of Virginia, North Carolina, and South Carolina (USA; NatureServe 2013). *Somatogyrus viriginicus* is an annual species, in which most adults die soon after reproducing (Johnson et al. 2013). Juveniles were collected on 6 August 2013 from a viable population in the Eno River (near Hillsborough, North Carolina, USA) and were immediately transported to our laboratory at North Carolina State University for testing. Average shell length, as measured from the top down, perpendicular to the spiral, was 1.84 mm (± 0.37 mm). Based on earlier sampling in the Eno River on 2 May 2013, in which only adults and eggs were found, the juveniles tested were <3 months old.

Experimental conditions

We selected herbicide treatment concentrations based on recommended application rates for treatment of hydrilla, maximum application rates reported on the product label, and acute toxicity data reported for other taxa in peer-reviewed literature (Crosby & Tucker 1966; Sanders 1969; Hamelink et al. 1986; Paul et al. 1994; Yi et al. 2011) and on Material Data Safety Sheets (SePRO Corporation 2009, 2010, 2011, 2012; UPI 2011, 2012). An analytically verified 1304 μg/L (parts per billion) stock solution of Sonar – PR® (fluridone) formulation was prepared and provided by the SePRO Research and Technology Campus (Whitakers, North Carolina, USA). An analytically verified stock solution of Sonar – Genesis® formulation was prepared at 1383 μg/L. The fluridone formulations were shipped via overnight courier to our laboratory at North Carolina State University and immediately refrigerated until use in toxicity tests. Acute test concentrations of fluridone formulations ranged from 2.5 to 200 μg/L with an additional treatment at the stock solution concentrations (1304 μg/L for PR® and 1383 μg/L for Genesis®). Concentrations of Sonar – PR® in the chronic (28-d) experiment ranged from 5 to 300 μg/L. A concentrated formulation of endothall (Aquathol-K®), labeled as 4.23 lb/gal (~506,866 mg/L), was hand delivered by collaborating personnel in the Department of Crop Science, North Carolina State University, and subsequently diluted to a working stock of 1000 mg/L (parts per million). Test concentrations of endothall ranged from 0.5 to 1000 mg/L. Composite water samples (10 mL from each of three replicates, 30 mL total volume) were collected for herbicide concentration verification prior to placing organisms into the chambers, and again at 48 h; samples were stored at 4 °C until they were shipped to SePRO analytical laboratory (fluridone quantified via HPLC) or the US Army Engineer Research and Development Center's Environmental Laboratory (endothall quantified via immunoassay; Gainesville, Florida, USA).

All experiments were static-renewal tests conducted in reconstituted soft water (ASTM 2007), with 90%−100% water renewal at 48 h in the 96-h acute non-aerated tests, and at 72-h intervals in the 28-d aerated experiment. Soft water was selected because it most closely approximated the water quality parameters in most of the test organisms' native ranges (e.g., Lake Waccamaw, Eno River). Quality assurance and control were ensured by conducting all tests according to the *Standard Guide for Conducting Laboratory Toxicity Tests with Freshwater Mussels* (ASTM 2006). No formalized guidelines exist for conducting experiments with freshwater snails or adult mussels, so the mussel guideline was used (ASTM 2006), as per protocol in other studies (Besser et al. 2009; Archambault et al. 2013). Organisms were acclimated from their culture water to the test water by placing them in a 50:50 solution of culture/reconstituted water for 2 h, then further diluting the culture water to a 25:75 ratio with reconstituted water, and held for an additional 2 h before being placed in 100% reconstituted water (ASTM 2006, 2007). Tests were conducted in light and temperature-controlled environmental chambers (Precision Model 818, Thermo Fisher Scientific, Marietta, Ohio, USA), held at 20 °C and LD 16:8. In the 24-h tests, ~150 glochidia were placed in each of three replicates per treatment. In the 96-h experiments, seven mussels or snails were placed in each of three replicates per treatment, with 10 organisms per replicate in controls (0 μg/L). Snails were transferred to untreated ASTM water at the conclusion of the test and held for 48 h for a post-exposure survival assessment to identify potential latent mortality effects (per J. Besser, 2013, e-mail to WGC; unreferenced). Mean water quality conditions among acute experiments were 27.8 mg $CaCO_3$/L alkalinity, 39.0 mg $CaCO_3$/L hardness, 220 μS/cm conductivity, 7.50 pH, and 8.47 mg/L dissolved oxygen ($n = 4$ for alkalinity and hardness, $n = 36$ for all other variables). In the 28-d experiment, five adult mussels were placed in each of three replicates per treatment. Mussels were fed a mixture of 1-mL Instant Algae® Shellfish Diet and 0.5-mL *Nannochloropsis* (Nanno 3600) concentrate diluted in 500 mL deionized water. Approximately, 6.25 mL of food mixture was added to each replicate (administered concentrations of 50,000 and 850,000 cells/mL solution, respectively; Reed Mariculture, Campbell, California, USA) every 72 h at least 2 h before each solution renewal (Mosher et al. 2012; Leonard 2013). Mean water quality conditions in the chronic experiment were 26.7 mg $CaCO_3$/L alkalinity, 44.9 mg $CaCO_3$/L hardness, 168 μS/cm conductivity, 7.49 pH, and 8.54 mg/L dissolved oxygen ($n = 9$ for alkalinity and hardness, $n = 54$ for pH, $n = 60$ for all other variables).

Data collection and statistical analysis

Viability was assessed at 24 h for a subsample of approximately 50 glochidia in each replicate. We assessed viability by exposing glochidia to a saturated NaCl solution and viewing them under a stereomicroscope; glochidia that exhibited a shell-closure response to salt were considered viable (ASTM 2006). At the end of each 96-h exposure, survival of juvenile mussels was assessed by viewing them under a stereomicroscope; juveniles that exhibited foot movement outside of the shell, foot movement inside the shell, or a detectable heartbeat within a five-minute observation period were considered alive (ASTM 2006). Snail survival was assessed similarly, by observing for righting or movement within five minutes. In the chronic experiment, survival of adult mussels was assessed visually every 72 h by observing for foot retraction or valve closure in response to dewatering during renewal in mussels with open shells. Because the shell of *L. fullerkati* is thin and fragile, we made no attempt to check for resistance to opening, and mussels with tightly closed shells were assumed to be alive.

The effects of herbicide concentration on the survival of mussels and snails were analyzed by using survival data to generate median lethal concentrations (LC50s) and 95% confidence intervals (CIs) via the Trimmed Spearman-Karber method (Comprehensive

Environmental Toxicity Information Software (CETIS)™, v1.8.0.12, Tidepool Scientific, LLC, McKinleyville, California, USA). The LC50 was defined as the concentration that caused mortality in 50% of the individuals in the exposed sample, and the LC05 was defined as the concentration that caused mortality in 5% of the sample. LC values were considered significantly different when their 95% CIs did not overlap (i.e., $\alpha = 0.05$).

In the 28-d experiment with adults, we made observations every 72 h of siphoning behavior and foot protrusion; mussels were given a binary designation of siphoning or not siphoning and assigned a binary score of foot protrusion or no foot protrusion (Leonard 2013). The effect of herbicide concentration on siphoning behavior and foot protrusion was analyzed using a repeated measures analysis of variance (PROC MIXED; SAS version 9.3; SAS Institute, Inc., Cary, North Carolina, USA). Significant effects ($\alpha = 0.05$) of fluridone concentration were further analyzed using a Dunnett's *post hoc* test.

Results

Herbicide concentration analysis

Exposure accuracy (i.e., measured herbicide concentration compared to target concentration) was calculated as: exposure accuracy = $(P_m)/(P_t) \bullet 100$, where P_m is the measured herbicide concentration and P_t is the target concentration. The measured concentration of the fluridone stock solution used in tests with mussels (Sonar – PR®) was 108.3% of the reported prepared concentration of 1304 μg/L, and the mean exposure accuracy in experiments was 119.9% (range 102%–176%) of target treatment concentrations. The verified concentration of the Sonar – Genesis® formulation at the time of testing with juvenile snails was 87.0% of the initial reported concentration of 1383 μg/L, and had a mean exposure accuracy of 85.2% (range 80%–102%) in treatments prepared from the stock. The mean exposure accuracy in endothall (Aquathol-K®) experiments was 109.0% (range 100%–114%) of target treatment concentrations.

Mussel toxicity

Control viability at 24 and 48 h in glochidia tests was >90% of the initial viability that was assessed on arrival to the laboratory for all experiments, in accordance with testing guidelines (ASTM 2006). Control survival in experiments with juveniles was >90%, except in the endothall experiment at the 96-h time point. Even though control survival (73.3%) at 96 h in the endothall experiment was slightly below the 80% recommended in the standard guideline for toxicity tests with juvenile mussels (ASTM 2006), results are reported herein because survival was >90% in three of the low concentration treatments (1, 5, and 10 mg/L).

The 24-h LC50 for *L. siliquoidea* glochidia exposed to fluridone (Sonar – PR®) was 865 μg/L (95% CI, 729–1026 μg/L) and the 48-h LC50 was 978 μg/L (787–1214 μg/L). The experiment with juveniles yielded a 48-h LC50 of 1197 μg/L (569–2522 μg/L) and a 96-h LC50 of 511 μg/L (309–843 μg/L; (Table 1)). The 24-h LC05 for glochidia was 290 μg/L (0–598 μg/L); LC05s in the juvenile tests and at the 48-h time point of the glochidia test could not be determined due to the lack of two or more partial mortality responses among treatments. A chronic LC50 could not be determined for *L. fullerkati* at any time point during the 28-d test because no mortality occurred. Adult mussels exhibited only minor foot protrusion behavior during the experiment. Moreover, the degree of foot extension observed was minimal. Initially, foot extension was recorded using four

Table 1. Median lethal concentrations (LC50s) for mussels and snails (with 95% CI) in acute and chronic exposures to herbicides commonly used to treat *Hydrilla verticillata*. ND = value could not be determined. 48-h post = post-exposure assessment.

Species	Life stage	Time point	Fluridone (μg/L)	Endothall (mg/L)
Lampsilis siliquoidea	Glochidia	24 h	865 (729−1026)	31.2 (30.3−32.2)
		48 h	978 (787−1214)	27.6 (25.5−29.9)
	Juvenile	48 h	1197 (569−2252)	214 (134−342)
		96 h	511 (309−843)	34.4 (29.3−40.5)
Somatogyrus virginicus	Juvenile	48 h	ND	–
		96 h	500 (452−553)	–
		48-h post	409 (329−509)	–
Lampsilis fullerkati	Adult	28 d	ND − no mortality	–

categories: (1) shell closed and/or foot not visible; (2) foot visible, but not extended beyond mantle margin; (3) foot extended beyond mantle; and (4) foot extended and swollen. Because observations (n = 1050) were recorded as category 1 (67.5% of observations) or 2 (32.4%) in all but one case, foot protrusion data were analyzed as a binary function (i.e., foot extended or not) like the siphoning data. A category 3 observation was recorded only once, and category 4 was never observed. We found no statistically significant effect of fluridone concentration on foot protrusion ($p = 0.06$) or siphoning behavior ($p = 0.08$).

In endothall exposures, the glochidia 24-h LC50 was 31.2 mg/L (30.3−32.2 mg/L), and the 48-h LC50 was 27.6 mg/L (25.5−29.9 mg/L). The experiment with juvenile *L. siliquoidea* yielded a 48-h LC50 of 214 mg/L (134−342 mg/L) and a 96-h LC50 of 34.4 mg/L (29.3−40.5 mg/L; (Table 1)). The 48-h LC05 for juveniles was 34.6 mg/L (3.90−80.0 mg/L); other LC05s were not determined due to the lack of two or more partial mortality responses among treatments or poor fit.

Snail toxicity

The 96-h LC50 for *S. virginicus* exposed to fluridone (Sonar − Genesis®) was 500 μg/L (452−553 μg/L) and the LC50 at 48-h post exposure was 409 μg/L (329−509 μg/L; (Table 1)). The overlapping CIs between the two assessment time points indicate that there was no significant latent mortality in the exposed snails ($\alpha = 0.05$). An LC50 at the 48-h time point and LC05s could not be determined due to the lack of two or more partial mortality responses among treatments.

Discussion

Our results indicate that the early life stages of *L. siliquoidea* are more acutely sensitive to fluridone than most other aquatic organisms that have been tested. In a multi-laboratory study evaluating the effects technical grade fluridone (i.e., active ingredient only) and a commercial formulation of Sonar® on freshwater and marine invertebrates and fishes, Hamelink et al. (1986) reported a mean LC50 of 4.3 mg/L for invertebrates (n = 15 tests among six species) and a mean LC50 of 10.4 mg/L for fishes (n = 28 tests among five species) (Table 2). By comparison, at 24 h, *L. siliquoidea* glochidia were approximately five times more sensitive than invertebrates they tested and 12 times more sensitive than

Table 2. Comparative acute toxicities of freshwater species exposed to technical or commercial formulations of fluridone and dipotassium salt of endothall.

Species	Common name	Chemical grade	Time point	LC50 (mg/L)	Source
		FLURIDONE			
Invertebrates					
Arrenurus sp.	A water mite	Formulation	96 h	0.010	Yi et al. (2011)
		Technical	48 h	0.891	Yi et al. (2011)
		Technical	96 h	0.631	Yi et al. (2011)
Chironomus plumosus	A midge	Formulation	48 h	1.3	Hamelink et al. (1986)
		Technical	48 h	1.3	Hamelink et al. 1986
Gammarus pseudolimnaeus	An amphipod	Technical	96 h	2.1−4.1	Hamelink et al. 1986
		Formulation	96 h	>32	Hamelink et al. (1986)
Daphnia magna	A water flea	Formulation	48 h	3.6−3.9	Hamelink et al. (1986)
		Technical	48 h	3.9−6.3	Hamelink et al. (1986)
Fishes					
Sander vitreus	Walleye	Formulation	96 h	1.8	Paul et al. (1994)
Oncorhyncus mykiss	Rainbow trout	Technical	96 h	4.2−11.7	Hamelink et al. (1986)
		Formulation	96 h	7.1−8.1	Hamelink et al. (1986)
Ictalurus punctatus	Channel catfish	Technical	96 h	8.2−15.0	Hamelink et al. (1986)
		Formulation	96 h	13.2	Hamelink et al. (1986)
Micropterus dolomieu	Smallmouth bass	Formulation	96 h	7.6	Paul et al. (1994)
Lepomis macrochirus	Bluegill sunfish	Formulation	96 h	12.0	Hamelink et al. (1986)
		Technical	96 h	12.1−13.0	Hamelink et al. (1986)
Micropterus salmoides	Largemouth bass	Formulation	96 h	13	Paul et al. (1994)
Pimephales promelas	Fathead minnow	Technical	96 h	22	Hamelink et al. (1986)

(*continued*)

Table 2. (*Continued*)

Species	Common name	Chemical grade	Time point	LC50 (mg/L)	Source
		ENDOTHALL			
Invertebrates					
Daphnia magna	A water flea	Technical	26 h (IC50)	46	Crosby and Tucker (1966)
Gammarus lacustris	An amphipod	Formulation	24 h	>100	Sanders (1969)
Fishes					
Sander vitreus	Walleye (8–10 d)	Formulation	96 h	16	Paul et al. (1994)
	Walleye (41–43 d)	Formulation	96 h	54	Paul et al. (1994)
Micropterus dolomieu	Smallmouth bass (<1 d)	Formulation	96 h	47	Paul et al. (1994)
Micropterus salmoides	Largemouth bass (9–13 d)	Formulation	96 h	130	Paul et al. (1994)

fishes, and juveniles at 96 h were approximately eight times more sensitive than other invertebrates and 20 times more sensitive than fishes. The closest relative to *L. siliquoidea* in their study was the eastern oyster (*Crassostrea virginica*); oyster embryos had a 48-h LC50 of 6.8 mg/L. *Lampsilis siliquoidea* was approximately 8 (24-h glochidia LC50) to 13 (96-h juvenile LC50) times more sensitive than oyster embryos. Another study determined the 96-h LC50s of fluridone for the early life stages of walleye (*Sander vitreus*), smallmouth bass (*Micropterus dolomieu*), and largemouth bass (*M. salmoides*) (Paul et al. 1994; Table 2), which were all more tolerant than the mussels tested here (Table 1). In a recent investigation of the toxicity of fluridone on male water mites (*Arrenurus* sp.), Yi et al. (2011) reported toxicities to technical grade fluridone similar to our Sonar − PR® commercial formulation results; however, they found water mites were 60 times more sensitive in tests with another commercial formulation (Sonar − AS®; Table 2).

In the context of typical treatment prescriptions for hydrilla, all of the mussel toxicity data generated in tests with Sonar® PR, including those generally reported for regulatory purposes (24 h for glochidia, 96 h for juveniles), are two or more orders of magnitude greater than the water column treatment maximum target concentration for Lake Waccamaw (5 μg/L), and are more than three times higher than the maximum label application rate of 150 μg/L (SePRO Corporation 2012).

As with fluridone, freshwater mussels were also more acutely sensitive to endothall than most other tested organisms. Median effective concentrations (EC50s) and LC50s for 11 species range from >100 to 1071 mg/L; channel catfish (*Ictalurus punctatus*) and coho salmon (*Oncorhynchus kisutch*) were the most sensitive in the group, and the bluegill sunfish (*Lepomis macrochirus*) was the most tolerant (UPI 2012). The nearest relative to unionid mussels included in the ecotoxicity data was the eastern oyster, which had a 96-h sublethal EC50 (shell deposition) of 335 mg/L (UPI 2012), approximately 10 times greater than our 24-h glochidia and 96-h juvenile LC50s for *L. siliquoidea*. Acute values reported for some species in other studies showed sensitivities more similar to those of unionid mussels, including early life stage smallmouth bass (aged <1 d) and walleye fry (aged 41−43 d; Paul et al. 1994), and *Daphnia magna* (26-h median immobilization concentration (IC50); Crosby & Tucker 1966) (Table 2). Walleye 8−10 days old (96-h LC50 = 16 mg/L; Paul et al. 1994) were approximately twice as sensitive as the *L. siliquoidea* tested here (Tables 1 and 2). There was good agreement in our data among the LC50s for glochidia and the 96-h juvenile LC50, suggesting a defined threshold of tolerance; most mussels survived at concentrations ≤10 mg/L and experienced complete mortality at concentrations ≥100 mg/L. Despite being among the most sensitive species tested to date, the toxicity data for *L. siliquoidea* are 6−34 times higher than the recommended application rate of endothall for treatment of hydrilla (1−5 mg/L; UPI 2011). The 24-h glochidia and 96-h juvenile LC50s are approximately one order of magnitude greater than the application rate, indicating a smaller margin of error in applying endothall compared to fluridone. It should be stressed that an LC50 is not protective of a population (i.e., only 50% are expected to survive at the LC50 concentration).

We did not find any significant effects of fluridone on lethal or sublethal endpoints in tests with *L. fullerkati*, suggesting that adult mussels were tolerant to the range of concentrations used over 28 d, and may be tolerant to seasonal exposures at 5 μg/L during treatment of hydrilla infestations. However, many other endpoints could be explored, and some may provide more insight into effects from chronic exposure. Relevant toxicological endpoints in sublethal studies of freshwater mussel sensitivity to other contaminants that may be applied in future fluridone and endothall studies include growth (in juveniles, Bringolf et al. 2007; Wang et al. 2007, 2011, 2013), glochidial metamorphosis success

(Hazelton et al. 2013), female mantle lure display (Bringolf et al. 2010; Hazelton et al. 2013; Leonard 2013), hemolymph and tissue analysis (Archambault et al. 2013; Leonard 2013), movement and burrowing (Flynn & Spellman 2009; Archambault et al. 2013; Hazelton et al. 2014), and metabolomics (Leonard 2013). We attempted to evaluate female mantle lure display in our experiment, but we had few females per replicate, thus there was insufficient statistical power to make sound inferences. We did note, however, that mussels in all treatments except for the highest concentration (300 μg/L) were periodically observed displaying mantle lures (stage 3 or higher, as per Bringolf et al. 2010). Our observations suggest that fluridone applied at a typically prescribed rate of 5 μg/L may not affect unionid mantle lure display. However, more statistically robust experimentation is needed to confirm a lack of effect, and to elucidate any other potential reproductive effects of fluridone.

The freshwater snail, *S. viriginicus*, was equally sensitive to the fluridone formulation Sonar – Genesis® as juvenile mussels were to the Sonar – PR® formulation (Table 1), and much more sensitive than other animals previously tested in commercial formulations of Sonar® (Hamelink et al. 1986; Paul et al. 1994), except for water mites (Yi et al. 2011) (Table 2). The reported acute values for Sonar – Genesis® were 1.8 mg/L (96-h LC50) for walleye and 3.6 mg/L (48-h EC50) for *Daphnia* (SePRO Corporation 2011), which are values 3.6 to 7.2-fold higher than the snail 96-h LC50. In experiments with *S. viriginicus*, both the 96 and 48-h post exposure LC50s were approximately two orders of magnitude higher than typical treatment concentrations recommended for hydrilla, and more than three times higher than the maximum label rate of application (SePRO Corporation 2010). Moreover, adult snails suffered no mortality in previous tests in our laboratory that had a maximum treatment concentration of 500 μg/L ((96-h LC50 > 500 μg/L), Archambault, Bergeron, and Cope, unpublished data). However, caution should be used in interpreting acute duration data, because slow-release or slow-acting herbicides like fluridone typically require extended exposure when treating hydrilla. Further, whole life cycle studies are especially important for *S. viriginicus* and other species that have an annual reproductive ecology, where the timing of hydrilla and other weed growth – and therefore herbicide treatment – coincides with egg laying, juvenile hatching and growth, and adult senescence. Because *S. viriginicus* adults die after reproduction (Johnson et al. 2013), negative effects to one cohort could result in further species decline.

In summary, we found that the fluridone and endothall concentrations typically recommended for hydrilla treatment were not acutely toxic to the freshwater molluscs tested in this study, and a 28-d exposure to fluridone was not lethal to adult mussels even at the highest concentration, indicating minimal risk of short-term effects to non-target species, including several protected and rare species. We also found that freshwater molluscs were more sensitive to fluridone and endothall than most other species previously tested. The mussels and snails studied here represent hundreds of highly imperilled freshwater gastropods and unionids, and our findings may signal their greater sensitivity to herbicides than other species commonly studied in aquatic toxicity testing (e.g., *Daphnia* spp., *Hyalella* spp., fathead minnows (*Pimephales promelas*)). Furthermore, chemicals like fluridone and endothall are sometimes used in combination to increase effectiveness against aquatic weeds, and are rarely the only chemicals present in surface waters (i.e., aquatic contaminants). They also are typically applied over a longer duration than our test exposures. Though fluridone and endothall have been used for aquatic weed management for decades, more research is needed to elucidate any potential risk to less-studied non-target taxa, including molluscs, especially given hydrilla's encroachment into more systems across the country with high native biodiversity and endemism (e.g., Lake Waccamaw, Eno River). By providing a more thorough picture of the potential ecological risk associated with applying such herbicides for control of invasive aquatic weeds, resource managers can more confidently evaluate them as an option among other management choices (e.g., no treatment, grass carp control, and mechanical removal) and their associated risks. Topics warranting future study include acute exposures of endothall to snails; chronic exposures of juvenile mussels and snails to fluridone and endothall; evaluation of short- and long-term sublethal effects to juvenile and adult molluscs (e.g., reproduction, transformation success, growth, and biomarkers); indirect effects (e.g., effects on diet/food availability); whole life cycle exposures; and multi-stressor studies.

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Environmental Assessment
For Controlling the Growth and Spread of
a Noxious Aquatic Weed, *Hydrilla verticillata*,
at Lake Waccamaw, North Carolina

This document was drafted by multiple authors and incorporates considerable input from the Lake Waccamaw Hydrilla Management Technical Advisory Group (Appendix A).

Edited by:
Rob Emens, NCDENR - Division of Water Resources. 919-707-9012
Ed Corey, NCDENR - Division of Parks & Recreation. 919-841-4037

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A. Proposed Project Description

1. History

Lake Waccamaw State Park consists of 10,694 acres in Columbus County, including an almost 9,000-acre natural lake, the third largest in the state and one of the most unique aquatic resources in the country. Lake Waccamaw is the largest bay lake in the region, and while all Carolina bays are unusual, this lake stands apart. Many Carolina bays are small, ranging about 500 feet in length; Lake Waccamaw has 14 miles of shoreline. Many bays are also totally dependent on rainfall, but Lake Waccamaw gets its water supply from the Friar Swamp drainage. Most Carolina bays also have naturally high levels of tannins, lending a low pH and a usual paucity of aquatic life. Limestone bluffs along Lake Waccamaw's north shore and a partial limestone lakebed neutralize the lake's water, however, making it suitable for many species of plants and animals, including species such as the Waccamaw killifish (*Fundulus waccamensis*), the Waccamaw darter (*Etheostoma perlongum*), the federally threatened Waccamaw silverside (*Menidia extensa*), Waccamaw fatmucket (*Lampsilis fullerkati*), Waccamaw snail (*Amnicola sp. 1*) and Waccamaw siltsnail (*Cincinnatia sp. 1*).

Lake Waccamaw State Park is an important regional recreation resource. The park offers a peaceful, natural setting for varied recreational activities such as hiking, camping, picnicking, fishing, and nature study. The sandy lake bottom and clear water are attractive for wading and swimming, and nearby ramps provide access for boaters. Access to the lake and Waccamaw River is also provided at the dam. Recreational activities at the park are appropriately limited to preserve the quality of the recreational experience and to protect the park's outstanding natural resources.

Lake residents and visitors noticed excessive growths of aquatic vegetation during the summer of 2012. Some of these citizens became concerned and contacted the Lake Waccamaw State Park (LAWA) and the NC Division of Water Resources (DWR) about their observations. Responding to these reports Rob Emens, DWR Aquatic Weed Specialist, arranged to meet with LAWA Ranger Jonathan Short to investigate. During the site visit on October 11, 2012, Emens identified monoecious hydrilla (*Hydrilla verticillata*) in the northwestern corner of the lake, near the NC Wildlife Resources Commission (WRC) boat ramp.

Hydrilla is a federal and state noxious weed, and is considered by many to be the perfect aquatic invasive plant (Langeland 1996). This species can grow in most types of soil and can survive in low light conditions. As Lake Waccamaw is relatively shallow (maximum depth is approximately 11 feet), there is no spot in the lake that could not be colonized by this plant. It can form extremely dense stands, filling the water column from the bottom to the surface, crowding out native vegetation and reducing habitat for freshwater mussels and other aquatic organisms. These mats can also affect recreation, including boating and fishing.

In November 2012, at the request of LAWA, DWR, and WRC, Dr. Rob Richardson, an associate professor and research specialist with the Crop Science Department at NC State University (NCSU) was contracted to do a systematic lake survey. The objective was to delimit the hydrilla infestation and document the presence and distribution of other aquatic plants. Dr. Richardson's team surveyed 362 points throughout the lake, using a combination of boat-mounted SONAR and vegetation sampling with a double-sided rake. All aquatic vegetation was identified to species and density was measured (subjectively) as a percent of cover of the sampling rake. Of the points surveyed, 279 (77%) contained native aquatic vegetation. Hydrilla was observed at 45 points (12%), and of those, 20 (5.5%) had 5% or greater coverage of the survey rake. Using different factors, including hydrilla presence, percent of cover, and biovolume (the height of the vegetation in the water column), Dr. Richardson's survey crew estimated 608 acres of the lake to be infested with hydrilla.

2. Proposed Action

LAWA Superintendent Toby Hall called forth a meeting of interested stakeholders to discuss the findings and derive a solution to the hydrilla issue at Lake Waccamaw. Representatives from the DWR, WRC, NCSU, Division of Water Quality (DWQ), Natural Heritage Program (NHP), NC Department of Agriculture (NCDA), SePRO, Winyah Rivers Foundation, and UNC-Wilmington, among others met on November 29th to discuss different plans of action. Dr. Richardson presented the results of a model of the likely spread of hydrilla if left untreated, based on habitat conditions in the lake and the behavior of the species elsewhere. The model indicates a rapid spread of hydrilla if unchecked, filling most of the lake in 5 years. Hydrilla would more than double its acreage in 2013, and would continue growing if unchecked (see Exhibit 1).

After examining several management options, including lake draining, harvesting, and biological control using sterile triploid grass carp (*Ctenopharyngodon idella*), the final, nearly unanimous decision was to move forward with an herbicide treatment for hydrilla in the lake. Considering the results of the survey, Dr. Richardson recommended treating one contiguous area (959 acres, Figure 1) with granular formulations of the chemical, fluridone. These areas, plus any additional areas that are found to be infested with hydrilla, will be treated annually to prevent the further growth and reproduction of this noxious weed, with the understanding that it will likely take up to 10 years of consecutive treatments to remove this species.

Figure 1. Map showing locations of hydrilla (shaded red) within Lake Waccamaw. Survey performed in November, 2012.

3. Methods

Successful management of submersed aquatic weeds is dependent on accurate field data and the experience of those who make the management decisions. Unlike managing terrestrial weeds and immersed aquatic vegetation, submersed aquatic vegetation (SAV) is much more difficult to detect and treat. Detection of SAV can be done by visual reconnaissance if water clarity allows. A common method used to detect and sample SAV is the use of rake, grapple, or similar weighted tool that will harvest SAV from the bottom. An efficient method of detecting the presence, and biovolume, of SAV is a SONAR device. Using a SONAR device, especially one that records (a.k.a. recording fathometer), in combination with sampling to confirm species identification, is the best way to map SAV. Mapping hydrilla is crucial to determining the treatment areas. Lake surveys to detect and map hydrilla will continue throughout this project (~10 years).

Once a treatment area has been identified, factors such as size, water depth, and water movement need to be considered. Unlike spraying terrestrial weeds, treating SAV requires attaining a concentration of herbicide in the water surrounding the target vegetation. Additionally, there is a minimum contact time that is needed in order for the herbicide to be effective. This is known as Concentration and Exposure Time (CET). The CET varies depending on the aquatic-use herbicide product and the target species. (Getsinger 1989).

For this project multiple herbicide products will be selected from to best fit each treatment area. In order to maximize efficacy, limit water-use restrictions, and mitigate damage to native plants, pelletized formulations of fluridone will be selected to treat large areas (particularly areas >10 acres). Areas that are smaller than 10 acres will be considered on a case-by-case basis and treated with products that balance efficacy, cost, and selectivity. Endothall is the active ingredient most likely to be used in these areas. However, other aquatic use herbicides may be selected if those would be anticipated to produce superior results under the specific conditions present.

Herbicide products will be applied by licensed pesticide operators using specialized equipment. The application work will be done with the use of boats; no aerial applications are being considered.

The timing of herbicide applications is important. The biological activity of herbicide compounds has a half-life. There are windows of opportunity when applications must be done such that herbicidal activity corresponds with the timing of plant growth. Initial applications will occur in response to hydrilla growth. Hydrilla tubers in Lake Waccamaw are expected to begin sprouting in the April-June timeframe. Fluridone, a systemic herbicide, requires a CET of 90-180 days; therefore, supplemental or "bump" applications will be required during the growing season to maintain concentrations of active herbicide in the treatment areas. Water samples taken from within the treatment areas and outside of the treatment areas will be analyzed for concentration of fluridone. That data will be used to fine-tune application rates.

B. Purpose and Need for Proposed Project

1. Hydrilla Introduction

Hydrilla is an aquatic perennial plant indigenous to Asia. A dioecious form was introduced to Florida in the 1960's. Within a decade it spread throughout Florida and since found its way to all other Southeast states. Dioecious hydrilla, which has male and female flowers on different plants, occurs in the southeast U.S. from South Carolina southward and west to Texas. A monoecious form of hydrilla, which has both male and female flowers on the same plant, has infested waters in North Carolina and northward into New England. It is likely that monoecious hydrilla was first introduced to the area in the mid-1970's, approximately the same time as this species was introduced near the Potomac River.

Monoecious hydrilla was first documented in North Carolina in 1980 at William B. Umstead State Park, near Raleigh. State officials, responding to this occurrence, inspected all surrounding lakes and found several other locations within Wake County (Neuse River Basin) to be infested. Management efforts commenced at all sites found during this initial survey, and hydrilla was successfully extirpated from nearly all sites. Within 20 years hydrilla found its way into other major basins (Roanoke and Catawba), yet remained generally limited to Piedmont waterways. However, over the last decade this noxious weed has spread into the Coastal Plain and Mountain regions of the state. Hydrilla continues to colonize new habitats. It now infests areas with flowing water (Cheoah River, Eno River, and Contentnea Creek), and in 2010, was even observed in the western reach of the Albemarle Sound. Hydrilla will grow in shallow water and to depths of 15 feet or more.

The largest infestation in NC is at Lake Gaston, where it persists in approximately 3,000 acres (with a potential for 8,000 total acres) of this 20,000-acre reservoir. Hydrilla will grow in shallow water and to depths of 15 feet or more.

Hydrilla reproduces via vegetative fragments, tubers (formed at the end of rhizomes), turions (formed at the leaf axils) and seed. Tubers can remain viable in the hydrosoil for seven years or longer (Dr. Rob Richardson, personal communication). Reproduction from seed is of minor importance compared to reproduction by vegetative methods, but seed may be an important mechanism for long-distance transport via the gut of waterfowl (Langeland 1996).

A 1984 report by the Lake Waccamaw Water Quality Committee (NC Dept. of Natural Resources and Community Development 1984) noted the significant threat posed by the potential introduction of hydrilla, as well as the potential for rooted aquatic vegetation to reach nuisance levels. A visual survey of aquatic vegetation in the fall of 1984 noted that maidencane (*Panicum hemitomon*), narrowleaf cowlily (*Nuphar sagittifolia)*, and American lotus-lily (*Nelumbo lutea*) appeared to be the dominant species, although southern naiad was also observed.

2. Invasive and Noxious Status

Hydrilla is a Federal Noxious Weed and by reference, therefore, also considered a Class A Noxious Weed in NC. Sale of hydrilla is prohibited and movement within the state is also prohibited without a permit. In addition, the NC Department of Environment and Natural Resources (NCDENR) recognizes hydrilla as a Noxious Aquatic Weed, thereby qualifying it for consideration of state cost-share funding for its control.

Hydrilla infestations impede water use and alter aquatic habitat. Major concerns include the disruption of water flow leading to flooding events, disruption of recreational activities (wading, swimming, boating, etc.) leading to economic loss, disruption of water withdrawal due to the clogging of intakes, stratification leading to anaerobic conditions, and habitat loss.

Hydrilla can harbor mosquitoes and impose public health issues due to the threat of mosquito-borne diseases. West Nile Virus and arboviral encephalitides are potential concerns, as are Dengue fever and malaria (North Shore Mosquito Abatement District 2005-2011).

Hydrilla has been linked to Avian Vacuolar Myelinopathy (AVM), a syndrome that results in death for American coots (*Fulica americana*) and other waterfowl, as well as the birds that prey on weak and dying individuals, including the bald eagle (*Haliaeetus leucocephalus*). Studies have identified hydrilla, among other invasive submersed aquatic vegetation, as primary supporting substrates for a toxin-producing cyanobacteria in the order Stigonematales. This cyanobacteria is not associated with native SAV. The consumption of aquatic vegetation harboring the cyanobacteria leads to AVM. There is a strong correlation between AVM cases and hydrilla infestations.

3. Benefits from Project

Lake Waccamaw, the largest Carolina bay and third largest natural lake in the state, is a unique water resource in North Carolina. Consequently, it is managed as a part of Lake Waccamaw State Park. It was officially designated as a North Carolina State Lake in 1929 under the State Lakes Act, and has been classified as an Outstanding Resource Water (ORW) in the NCDENR, Division of Water Quality classification system. The unique character of Lake Waccamaw cannot be sustained with a spreading infestation of hydrilla.

A limestone bluff along the north shore of the lake reduces the acidity levels of the lake water and creates an environment that is ideal for a wide range of aquatic plants and animals. The lake is critical habitat for the federally threatened Waccamaw silverside (*Menidia extensa*), an endemic fish that occurs only in the lake and the upper reaches of the Waccamaw River. Other species endemic to the lake include Waccamaw killifish (*Fundulus waccamensis*), Waccamaw darter (*Etheostoma perlongum*), Waccamaw fatmucket (*Lampsilis fullerkati*), Waccamaw siltsnail (*Cincinnatia sp. 1*), and

Waccamaw snail (*Amnicola sp. 1*). Control of hydrilla will prevent displacement of the native vegetation, including several state endangered and threatened species, which has provided habitat for these populations of aquatic species in the past.

The private shore line around Lake Waccamaw has a high real estate value and the lake is popular for boating, swimming and fishing. If hydrilla is not managed recreational activities in the lake will be highly degraded or even eliminated. This in turn will lead to a decrease in shore line real estate values. In 2008 a study was completed detailing the economic contributions of the Division Parks and Recreation to the citizens of North Carolina. The study found the Division contributes $289 million in sales, $120 million in resident's income and 4,924 full-time equivalent jobs. The contribution of the park systems to both the state and local economies is significant. Loss of revenue from impacts of hydrilla could have a significant negative impact on both revenue and land values adjacent to the lake (Greenwood and Vick 2008).

The Waccamaw River and nearby waterways, such as Big Creek, Friar Swamp and Cove Swamp are not infested with hydrilla. The closest known site to be infested with hydrilla is approximately 50 miles from Lake Waccamaw (west of Pembroke). Controlling the growth of hydrilla in Lake Waccamaw will help to prevent the spread of hydrilla into surrounding areas.

In summary, the benefits of this project include:

1) Preserving the unique qualities of a lake that qualified as a NC State Park and was awarded classification as "Outstanding Resource Water";
2) Reducing (and potentially eliminating) the negative impacts that uncontrolled hydrilla growth will likely impose on the lake's unique aquatic species, including endemic taxa;
3) Preserving property values adjacent to and near the lake;
4) Securing the continuation of high quality recreational pursuits including fishing, swimming and boating; and,
5) Correcting Lake Waccamaw so that it does not remain as a source of hydrilla contamination to the surrounding area.

C. Alternative Analysis

1. No Action

A "no action" response will allow uncontrolled spread of hydrilla both within the lake and in connecting water bodies. Fragmentation (i.e., fragments of plants drift to new areas, take root, and develop into complete plants) is the primary method hydrilla propagates and spreads within a water body. This form of asexual reproduction allows the plant to spread very quickly within an aquatic system. Hydrilla typically grows into dense stands which lead to increased cover and habitat for some fish. However, studies have shown that once a system is invaded hydrilla will continue to occupy more and more "real estate" over a number of years and out-compete native vegetation. If hydrilla was to develop into a monoculture it would lead to severe and possibly irreversible damage to the endemic species of Lake Waccamaw. Current estimated rate of spread indicate a 100% increase in area per year over the next three years. A delay in management (temporary "no action") would lead to an infestation that will be even more costly to begin managing with herbicide. In summary, the problems associated with hydrilla will multiply with uncontrolled spread – loss of recreational use, alteration of habitat (impacting fish, wildlife & native plants), and depreciation of adjacent real estate.

2. Biological Control

Biological control of hydrilla can be separated into three different categories: non-selective generalist herbivory, selective herbivory or selective pathogenicity (target weed-specific disease).

Non-selective Herbivory
With respect to generalist herbivory, sterile triploid grass carp (*Ctenopharyngodon idella*) have been shown to be a cost-effective, but non-selective option for hydrilla management (Webb et al. 1994, Hanlon et al. 2000, Bonar 2002). Unlike the common carp, grass carp are herbivores. They prefer herbaceous SAV like hydrilla and Southern naiad. In order to attain complete hydrilla removal from grass carp herbivory pressure a stocking regime would prescribe enough fish to remove all hydrilla and naiad from the system. Other vegetation, though less palatable, will be consumed in the absence of preferred vegetation. Past efforts indicate that grass carp should only be used as the primary hydrilla management tool when a complete removal of submersed (and some emergent) aquatic vegetation is an acceptable outcome.

Selective Herbivory
Selective herbivory through use of hydrilla host-specific insects has been a desired biocontrol approach for several decades. Out of four efforts since 1987 at insect introductions to attack hydrilla, only one species has successfully established in dioecious hydrilla in Florida (the leaf-mining fly *Hydrellia pakistanae*), and none have

proven to provide effective control of hydrilla populations (Hetrick and Langeland 2012). *Hydrellia pakistanae* was previously released on Lake Gaston, NC, but did not establish. It is currently believed that this species cannot overwinter in North Carolina because monoecious hydrilla shoot biomass dies off each winter. Dioecious hydrilla biomass is persistent in Florida, providing an overwinter habitat for the species.

Host-specific Pathogens
Various efforts have been made to develop host-specific pathogens for hydrilla management. Several efforts over the last three decades have been made to commercialize the fungal pathogen *Mycoleptodiscus terrestris* for use as an inundative bioherbicide for hydrilla control (example: Shearer et al. 2011). All bioherbicide development to date has been unsuccessful.

3. Chemical Control

There are currently seven EPA-approved herbicides with some operational use for control of hydrilla: fluridone (Sonar®), endothall (Aquathol®) copper, diquat (Reward®), penoxsulam (Galleon®), bispyribac (Tradewind®), and flumioxazin (Clipper®). The Florida Fish and Wildlife Conservation Commission has produced an excellent web-based information system on aquatic plant management in Florida (http://plants.ifas.ufl.edu /manage/why-manage-plants/floridas-most-invasive-plants/hydrilla). It synthesizes much of the current understanding of control methods for hydrilla including use of herbicides that generally translates well for consideration for management in NC and can be a good secondary resource beyond information provided here.

Fluridone has been used since 1986 for hydrilla control. Fluridone's bleaching mode of action provides gradual control of sensitive target weeds like hydrilla over typically 45 – 90 days depending on plant establishment. Hydrilla is highly sensitive to fluridone with 3 – 5 ppb able to control the plant with sustained exposure. Typical contact-type herbicide treatments of aquatic vegetation leave some remaining biomass of the target plant (typically partial root crowns). Fluridone, a systemic-type herbicide, will control all hydrilla biomass above the hydrosoil. This translates to longer-term control since recovery must come from tubers. Fluridone is highly effective on monoecious hydrilla with low-dose applications, particularly when starting treatment as tubers begin sprouting (late spring). This is when the plant's carbohydrate reserves are at their lowest. In eradication efforts for monoecious hydrilla, immediate injury and growth suppression of low-dose fluridone on sprouting plants prevents their re-establishment in the spring and prevents maturation of the plant. Maturing hydrilla plants lead to potential fragmentation and spread and eventually the formation of new tubers. Minimal water use restrictions with low doses allow hydrilla management with fluridone without disruption of water use (lack of restrictions on fishing, swimming, domestic use, and most forms of irrigation).

Endothall, diquat, copper, and flumioxazin are contact herbicides. Although all have different modes of herbicidal action, all are faster acting products than fluridone, with diquat, copper and flumioxazin providing very fast weed control within days while endothall may take several weeks for hydrilla knockdown. Hydrilla control from contact herbicides generally leaves all or part of the lower part of the plant, typically referred to as the root crown of the plant. Some contact herbicides, such as copper-based products, can be toxic to mollusks.

Recent research has indicated that longer exposures (1 – 2 weeks in cooler water) to lower doses of endothall (1 – 2 ppm) at larger scales can provide improved control of dioecious hydrilla in Florida. Endothall is typically the most selective of the contact herbicides if used at lower rates but commonly is not as selective as lower-dose treatment with fluridone.

The remaining products currently used on hydrilla are newly-registered ALS (acetolactate synthase inhibitor) herbicides. The ALS herbicides penoxsulam and bispyribac shut down hydrilla growth at use rates of 10 – 15 ppb for penoxsulam and 20 – 30 ppb for bispyribac. These herbicides will control hydrilla growth if contact exposure time can be maintained for extended periods (several months). Due to slow activity on established hydrilla biomass, ALS herbicides have been combined with endothall to provide both initial knockdown and extended hydrilla control. Another ALS herbicide, imazamox (Clearcast®), has hydrilla activity but is more commonly used as a selective growth regulator that can provide several months of strong suppression with short exposure (several days).

4. Mechanical Control

Cultivators and rotovators can be used in a manner similar to a garden tiller for controlling aquatic vegetation. This process is intensely disturbing of lake sediment and would cause great increases in lake turbidity. Benthic organisms, such as mollusks and macroinvertebrates, would be rotovated indiscriminately from plants.

Drags, such as cables and chains, may be pulled by boats or winches to pull aquatic plants out. This process creates large amounts of plant fragments that can spread the infestation, and can significantly disturb lake sediment. This technique is recommended for small areas and cannot be practically used to treat large bodies of water.

Dredging can be a very effective method of removing aquatic plants from certain sites. It can also provide long term control of plants that do not produce seed or vegetative propagules. However, dredging is extremely expensive, with costs as high as $6,000 per acre. It is not selective and will remove all organisms in the dredge path. Dredging is highly disturbing of lake sediments and will increase lake turbidity. Dredge spoils would also need to be disposed of away from water so hydrilla does not spread to other water bodies. Because hydrilla does produce tubers in the hydrosoil, it is unlikely that dredging would remove 100% of the hydrilla infestation and hydrilla would rapidly repopulate the lake. There is strong potential for negative impacts to other benthic organisms.

Mechanical cutters use a reciprocating cutting bar to clip off plant biomass underwater (McComas 2003). This process, however, creates enormous amounts of plant fragments which will disperse throughout the lake spreading the infestation.

Mechanical harvesting combines cutting with a mechanical collection of most clipped biomass. Harvesting has some limited utility for clearing hydrilla biomass out of heavy use areas, but more as a maintenance strategy than a control measure: there is no published research to indicate that mechanical harvesting can reduce hydrilla infestations over time. This process is expensive, providing no long-term control and often exacerbating problems by spreading plant fragments. Because 99% of the current hydrilla infestation is limited to the northwest portion of the lake, fragmentation of hydrilla by mechanical harvesting would greatly increase the likelihood of new infestations in other portions of the lake. Harvesting has shown few advances in technology in the last several decades (Haller 1996). Additionally, harvesting is non-selective across flora and fauna and can produce significant mortality of fish and other aquatic animals in the harvested zone (Haller et al. 1980, Haller 1996). Haller et. al (1980) estimated that 32% and 18% of total fish numbers and total fish weight, respectively, were removed as by-catch from each acre mechanically harvested. Harvesting is a slow process, with only 2 to 8 acres completed per day (USACE-ERDC 2013). That pace would require 76 to 308 days to harvest the known hydrilla infestations in Lake Waccamaw; a timeframe which would allow continued hydrilla spread and reproduction in the Lake, thus not offering control. Disposal sites away from water would also be required for the thousands of tons of hydrilla biomass removed.

Weed rollers are small scale devices for installation around objects like docks (McComas 2003). Mechanical rollers travel back and forth over the lake sediment disturbing plants and other organisms in the travel path. Rollers work best on sandy sediments and may not work on mucky bottoms.

5. Physical Control

Physical control methods, including hand removal, benthic barriers, and lake-level drawdowns, have been used to control hydrilla.

Hand removal is the most common form of management of hydrilla, and can prove effective in very small areas of infestation, such as around intake valves and boat docks. However, this method is not considered cost-effective for large-scale areas of invasive plants. Aquatic plants may be up to 98% water, creating back-breaking labor, and these plants often reproduce as fast as they are removed from the area. Additionally, the fragments and tubers left behind can actually increase the target area. To evaluate the feasibility of controlling hydrilla by hand removal, the Eno River State

Park coordinated organized events during the summer of 2011. A section (approximately 100' in length) of the Eno River that contained hydrilla was selected. This area falls between Fews Ford and the suspension bridge. During June-July a cumulative time equaling 290 man-hours by volunteers and an additional 20 man-hours from park staff were spent pulling hydrilla from the Eno River. This effort resulted in removal of 85-90% of the visible biomass in the upper ½ of the 100' section and ~40% of the visible biomass removed from the lower ½ of the section. A final session of hand removal effort conducted by volunteers occurred in early August. In late September a visual inspection of the site reported no significant difference in density of hydrilla biomass within project area compared to control sites (sections immediately upstream and downstream of project site). The project was considered unsuccessful, and discontinued. (Keith Nealson, personal communication)

Benthic barriers are large mats laid down over an area to prevent light reaching the bottom. They can prove effective in broad spectrum control of aquatic plants, and have an immediate effect, with continued use over several years. Unfortunately, these barriers are only effective on a small scale, and can be quite expensive (upwards of $3,000 an acre). This method is non-selective, and can have severe impacts on non-target organisms, including mussels and spawning fish.

The drawdown of water in a treatment area can have certain advantages. It can be very effective on many species, with little-to-no cost per acre, and can be used in conjunction with other treatment options. Historically, Lake Waccamaw would have experienced periodic natural drawdowns, before the building of the dam at the Waccamaw River. This lack of water would have stimulated germination or sprouting of native plant species. Unfortunately, human-induced drawdowns are not very selective in their effects and can negatively impact recreational use of the lake, such as boating, fishing, and swimming during the drawdown. The timing of a drawdown to be effective for controlling hydrilla would begin in June and end in September. To control all hydrilla in the lake all lake-bottom areas that contain tubers would need to be dewatered. This would involve a significant drawdown since hydrilla has already produced a tuber bank in water depth of 5'.

6. Regulatory Control

Regulatory control of aquatic invasive species (AIS) typically focuses upon education and enforcement policies designed to prevent spread to non-infested areas. A number of national public awareness campaigns portray the negative environmental and economic impacts of AIS, including hydrilla. Signage posted at public boat ramps target those specific users. Educating the community at large about what AIS are, how they spread, the problems associated with their establishment, and what people can do to prevent the introduction/movement of AIS should be messages that are transmitted through multiple forms of media. Newspapers local to Lake Waccamaw have already printed articles about the hydrilla infestation. Continued outreach and press is needed to further educate the community about AIS.

One regulatory option that would limit the inadvertent anthropogenic movement of hydrilla from Lake Waccamaw to offsite locations would be to close or aggressively restrict use of the public boat access areas. Hydrilla fragments readily contaminate equipment like boats, boat trailers, and boat motors/props. These types of equipment are known vectors whereas hydrilla "hitchhikes" on these as they are moved from one location to another. Restricting the egress of equipment from the lake would be the only immediate method of limiting and/or reducing this vector.

Cleaning stations have been designed specifically to decontaminate boats/equipment of AIS. Once appropriate decontamination systems are in place the public boat access areas can be re-opened. Another option to re-open access during limited hours would be to retain a dock-master trained in the inspection and removal of AIS. Each boat entering and leaving Lake Waccamaw would be required to undergo an inspection for AIS by the dock-master.

7. Preferred Alternative

After careful consideration of each control option, the Lake Waccamaw Hydrilla Management Technical Advisory Committee has decided to pursue chemical treatment, with fluridone and endothall as the primary chemicals. In terms of hydrilla management objectives for Lake Waccamaw fluridone has use properties and a proven nationwide track record for ecologically compatible use in eradication efforts. Through use of one or more EPA-registered, controlled release fluridone pellet formulations (such as Sonar SRP, PR, One, or Q), a partial-lake treatment program for hydrilla control and tuber bank attrition is feasible. Recently in NC, a sustained six-year program of pelletized fluridone use in the Tar River Reservoir has successfully removed over 99% of hydrilla tubers from an infested section of the reservoir. Small isolated patches of hydrilla on Lake Waccamaw subject to high dilution potential and considered too small for optimal use of fluridone will require the use of contact herbicide(s).

Public awareness actions will be taken, including notifying boaters of the presence of this noxious aquatic weed through appropriate signage and kiosks at boat ramps. Education efforts targeting local user groups, such as home owners, fishermen and vacationers will focus on the development of cultural habits that ultimately reduce the frequency that AIS get transported. See Habitatitude (http://habitattitude.net/) and Protect Your Waters (http://www.protectyourwaters.net/) for more information and examples of preferred practices.

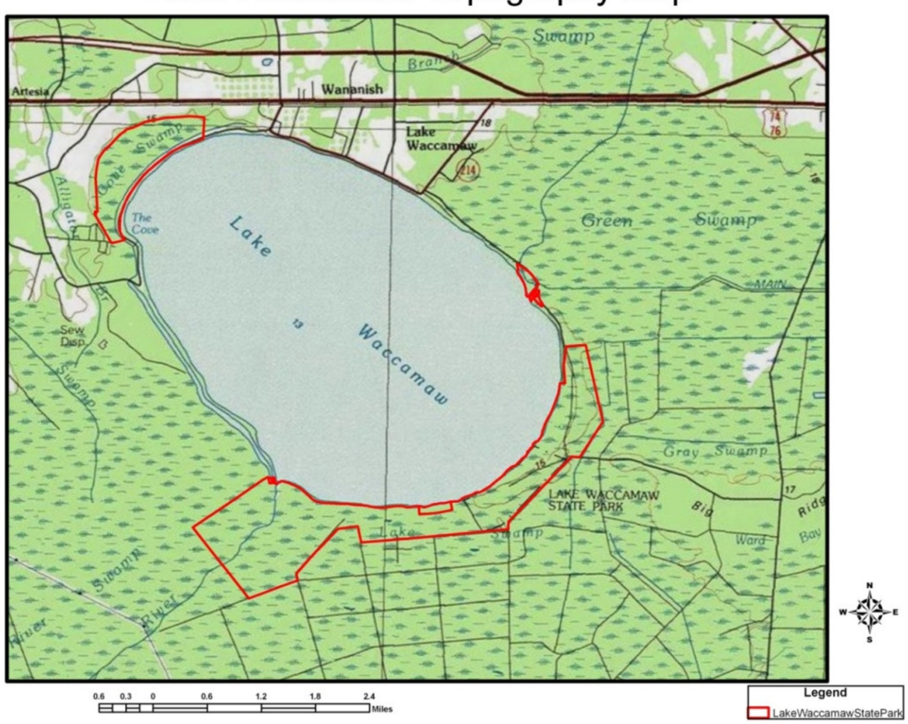
A "No Action" option would allow uncontrolled spread of hydrilla, further jeopardizing native unique and endemic species of the lake. Biological control using triploid grass carp would lead to an overall decline in native vegetation due to indiscriminate feeding on hydrilla and the abundant native naiad species. Mechanical control methods are non-selective, expensive, and have a high potential of injuring/removing native aquatic species. The size of the hydrilla infestation precludes physical control methods such as hand removal (which has been shown to be ineffective in other efforts in the state) and benthic barriers/covers, which are too expensive and non-selective to be deemed appropriate. Benthic barriers and would negatively impact unique and endemic aquatic life in the lake. Furthermore, lake drawdown would be inappropriate, as it would impact many user groups around the lake, including land owners, boaters, fishermen, and swimmers.

D. Existing Environmental Characteristics of Project Area

1. Topography

Lake Waccamaw is a Carolina bay and the largest bay lake in North Carolina. The dimensions of the lake are 5 miles long by roughly 3.3 miles wide. The lake encompasses roughly 8,938 acres. The lake is shallow, with a mean depth of 4.9 ft and a maximum depth of 10.8 feet. The lake has 14 miles of shoreline. The land that surrounds the lake is typically flat and swampy, except for the limestone bluffs located on the northeastern shore. There are also two sand ridges near the southeastern shore, a feature typical of most bay lakes. Five swamps are located around the lake, including Cove Swamp, Green Swamp, Lake Swamp, Friar Swamp, and Bogue Swamp. Elevation of the area ranges between 45 and 60 ft. Substrate throughout the lake is mostly sandy. The lake is fed by Big Creek which runs along the northeast shoreline. The main outlet of the lake is the Waccamaw River.

Lake Waccamaw Topography Map



2. Soils

Hydrilla is limited to permanent aquatic habitats. The U.S. Department of Agriculture, Natural Resources Conservation Service does not classify permanent water bodies as a soil series.

3. Land Use

Land uses surrounding the lake are predominately residential, single family homes, and forested park land. The park is predominately used for recreation. A few commercial uses are on the immediate shore including a campground, restaurant, and sailing club but they represent a small percentage of the total use.

4. Wetlands

Much of the shoreline of Lake Waccamaw is lined with palustrine wetlands. The unaltered wetlands along the shore of Lake Waccamaw State Park include an open stand of both bald-cypress and pond-cypress (*Taxodium distichum* and *Taxodium ascendens*). Beds of maidencane and twig-rush (*Cladium mariscoides*) cover some areas, while a sandy drawdown zone supports a diverse mix of herbaceous plants that includes one-flower hardscale (*Sclerolepis uniflora*), coinleaf (*Centella asiatica*), spikerush (*Eleocharis olivacea* var. *olivacea*), eastern doll's-daisy (*Boltonia asteroides var. glastifolia*), and globe-fruited seedbox (*Ludwigia sphaerocarpa*). This wetland community is classified as the Natural Lake Shoreline Swamp (Lake Waccamaw Subtype), and exists nowhere other than at Lake Waccamaw (Schafale 2012). Shallow water along parts of the shoreline has a floating aquatic community dominated by narrowleaf pondlily. This is the only such community known in a lakeshore setting, and is classified as the Natural Lake Shoreline Marsh (Lake Waccamaw Pond-Lily Subtype) (Schafale 2012). The wetland communities on the Lake Waccamaw shoreline are crucial habitat for rare plants. A total of 11 rare plant species are associated with the shoreline wetlands or shallow water.

Additional wetlands are connected to Lake Waccamaw, including the vast Cypress-Gum Swamp along the Waccamaw River where it emerges from the lake, and in Friar Swamp, a tributary of the lake.

5. Prime or Unique Agricultural Lands

There are no prime or unique agricultural lands adjacent to Lake Waccamaw.

6. Public Lands and Scenic, Recreational, and State Natural Areas

With the State Lakes Act of 1929, 7 state-owned natural lakes over 50 acres in size (including Lake Waccamaw) would henceforth be "administered as provided for other recreational areas now owned by the" Division of Parks and Recreation, then the Department of Conservation and Development. This designation provides opportunity for recreation, while restricting commercial use, construction, and consumptive water usage, including irrigation.

In 1976, the first 273 acres of Lake Waccamaw State Park were purchased, building over the next few decades to the current size of 1,732 acres. All 34 state parks in North Carolina provide equal opportunities for recreation, conservation and education. Generally, State Parks are expected to possess both significant natural resource values and significant recreational values. State Parks are expected to accommodate the development of facilities, but may vary in the extent of development depending upon what can be provided without damage to the scenic or natural features.

President Obama's America's Great Outdoors (AGO) Initiative was launched in 2010 to establish a community-based, 21st century agenda for conservation, recreation, and reconnecting Americans to the outdoors. In November 2011, then-Secretary of the Interior Ken Salazar released a final 50-State America's Great Outdoors Report outlining more than 100 of the country's most promising projects designed to protect special places and increase access to outdoor spaces. The full report contained two projects per state and, for the state of North Carolina, the Waccamaw watershed was chosen for its unique natural characteristics and collaborative efforts to promote the Waccamaw River Blue Trail, using recreation as a tool to promote conservation. Subsequently in May 2012, the Waccamaw River was included in a list of river projects to serve as a model of the AGO Rivers Initiative to conserve and restore key rivers across the nation, expand outdoor recreational opportunities and support jobs in local communities. Federal, state and local agencies and non-governmental organizations are collaborating to obtain both a National Water Trail designation, with a goal of facilitating outdoor recreation, and a National Blueway designation, recognizing river systems conserved through diverse stakeholder partnerships that use a comprehensive watershed approach to resource stewardship, for the Waccamaw River Blue Trail.

7. Areas of Archaeological or Historical Value

There are some archaeological areas at Lake Waccamaw where Native American artifacts including dugout canoes, a pre-historic whale skull, and other remnants of marine fossils have been recovered from the lake. The fossils were found along the north shore of the lake. The fossils are embedded in limestone which is located in the lake bottom and also on the bluffs of the north shore. These fossils represent the pre-historic era when the ocean covered what is now called the Coastal Plain of Southeastern North Carolina. The dug-out canoes also represent the important historical culture of the Waccamaw Siouan Tribe who once inhabited the area.

8. Air Quality

The NC Division of Air Quality (DAQ) has no monitoring stations in Columbus County. The air quality within the county is assumed to be good. DAQ operates an air quality monitoring station in Robeson County. No other counties adjacent to Columbus are outfitted with DAQ stations. Nearby counties that do have monitoring stations include: Sampson, Duplin, and New Hanover. Monitoring and emissions data collected from those stations is available, contact DAQ.

9. Noise Levels

Existing noise at Lake Waccamaw is scant. No airports, military operations, or industry exist near the lake; it is a rural area.

10. Water Resources

Lake Waccamaw is in the Yadkin - Pee Dee River Basin, which extends across Virginia, North and South Carolina. The Lake lies within a portion of the Pee Dee River Basin that extends into North Carolina. This smaller sub-basin is typically referred to as the Lumber River Basin.

Lake Waccamaw is classified by the NC Division of Water Quality (DWQ) as B Sw ORW . The "B" classification is assigned to those waters where primary recreational activities (i.e. swimming, water skiing, and similar uses involving human body contact with water) take place in an organized manner or on a frequent basis. The "Sw" supplemental classification refers to Lake Waccamaw as "Swamp water". Sw is intended to recognize those waters which are topographically located so as to generally have low velocities and other natural characteristics which are different from adjacent streams draining land with steeper topography. Swamp waters have naturally lower pH and dissolved oxygen levels and therefore have different water quality standards than other North Carolina water bodies. In addition to the chemical/physical differences, swamp waters are also evaluated for biological communities using modified criteria.

The lake was classified as an Outstanding Resource Waters (ORW) in 2000. This classification is intended to protect unique and special waters having excellent water quality and being of exceptional state or national ecological or recreational significance. Lake Waccamaw is considered to be of national significance by the North Carolina Natural Heritage Program and provides high recreational and scenic value. It is an important component of the Lake Waccamaw State Park and the shoreline along the State Park is classified as a Unique Wetland, a supplemental classification for wetlands of exceptional state or national ecological significance. These wetlands may include areas that have been documented as habitat essential for the conservation of state or federally listed threatened or endangered species. The Lake has also been designated as Critical Habitat for the federally threatened Waccamaw silverside. This means that the Lake contains features that are essential for the conservation of the Waccamaw silverside and special management and protection measures may be required under the Endangered Species Act.

DWQ has monitored the lake through the Ambient Monitoring System (AMS) at a sampling station at the dam on the southwest end of the lake. For the most part, sampling has been conducted on a monthly basis since 1974. The DWQ Ambient Lakes Monitoring (ALM) program has been sampling the lake at three locations on a 5-year rotating cycle since 1981. The most recent assessment was in 2011, and according to the North Carolina Trophic State Index (NCTSI), the lake was rated as moderate to highly productive. This rating is consistent with previous NCTSI ratings for the lake. Based on the AMS and ALM monitoring results, in 2011 the lake was reported to be in good condition and considered to be supporting its designated uses for primary recreation and aquatic life, with the exception of mercury levels in fish. All surface waters in North Carolina are currently impaired for fish consumption due to elevated mercury levels. High mercury levels in fish tissue are a global problem, thought to result primarily from atmospheric deposition.

Since 2008 DWQ has responded to several complaints regarding nuisance growths of filamentous green and blue-green algae in the lake. Complaints were also received about extensive growth and large mats of southern naiad. In some cases the complaints were based on vegetation interfering with recreational activities (swimming & boating); other complaints were based on foul odors following die-off of algal mats. Drought and record high temperatures appear to have exacerbated nuisance conditions in 2008, 2011 and 2012. (Mark Vander Borgh, personal communication)

Lake Waccamaw is highly prized for its recreational opportunities; including boating, fishing, and swimming. In keeping with regulations associated with the State Lakes Act of 1929, the lake is not utilized as a water supply source.

11. Forest Resources

Aside from the lands comprising Lake Waccamaw State Park, there are no forest resources directly adjacent to the lake.

12. Shellfish or Fish and Their Habitats

Lake Waccamaw is unique because of its water chemistry and relatively large size. Other bay lakes in North Carolina are highly acidic, but because limestone underlies the lake and forms the bluff along the northern shore, the pH of the water is near neutral, supporting a diversity and abundance of aquatic life. Because of this unique habitat, there are many species of rare fish and mollusks that are present in Lake Waccamaw (Table 1, Lindquist and Yarbrough 1979, Porter 1985, Mottesi 1998, Shute et al. 2000).

Of over fifty species known from Lake Waccamaw and surrounding waters, three species of fish are known only from the lake and the Waccamaw River directly below the dam. The Waccamaw silverside is listed by the US Fish & Wildlife Service (USFWS) as threatened, and the entire lake and a short reach of lower Big Creek (USFWS 1993) are designated as Critical Habitat. Other endemic fishes include the Waccamaw darter and the Waccamaw killifish. All three of these endemic fishes are considered to be common in the lake.

Lake Waccamaw and its adjacent waters hold a variety of mollusks, including fifteen species of aquatic snails and at least fourteen freshwater mussels. The endemic species of mollusks in Lake Waccamaw include the Waccamaw fatmucket, Waccamaw snail, and the Waccamaw siltsnail.

Table 1. Rare aquatic species found in Lake Waccamaw.

Scientific Name	Common Name	Taxa	Federal Status	NC Status
Elassoma boehlkei	Carolina Pygmy Sunfish	fish	FSC	T
Enneacanthus obesus	Banded Sunfish	fish		SR
Etheostoma perlongum	Waccamaw Darter	fish	FSC	T
Fundulus chrysotus	Golden Topminnow	fish		SR
Fundulus waccamensis	Waccamaw Killifish	fish	FSC	SC
Menidia extensa	Waccamaw Silverside	fish	T	T
Noturus sp. 2	Broadtail Madtom	fish	FSC	SC
Elliptio folliculata	Pod Lance	mussel		SC
Elliptio waccamawensis	Waccamaw Spike	mussel	FSC	E
Lampsilis cariosa	Yellow Lampmussel	mussel	FSC	E
Lampsilis fullerkati	Waccamaw Fatmucket	mussel	FSC	T
Lampsilis radiata	Eastern Lampmussel	mussel		T
Lampsilis splendida	Rayed Pink Fatmucket	mussel		SR
Leptodea ochracea	Tidewater Mucket	mussel		T
Toxolasma pullus	Savannah Lilliput	mussel	FSC	E

Villosa delumbis	Eastern Creekshell	mussel	SR
Amnicola sp.1	Waccamaw Snail	snail	SC
Cincinnatia sp.1	Waccamaw Siltsnail	snail	SC
Ferrissia hendersoni	Blackwater Ancylid	snail	SC
Lioplax subcarinata	Ridged Lioplax	snail	Watch list
Viviparus intertextus	Rotund Mysterysnail	snail	SR
Procambarus ancylus	Coastal Plain Crayfish	crayfish	Watch List
Procambarus braswelli	Waccamaw Crayfish	crayfish	SC

13. Wildlife and Natural Vegetation

Wildlife

Many species of wildlife occur within ½ mile of the lake shore, including mammals, birds, reptiles, amphibians, and a variety of invertebrates. Several taxonomic groups have direct interactions with the waters of Lake Waccamaw. Northern river otters (*Lontra canadensis*) feed on the fish, shellfish and crustaceans found here, as do bald eagle, osprey (*Pandion haliaetus*), belted kingfisher (*Megaceryle alcyon*), several species of herons and egrets (family Ardeidae), and white ibis (*Eudocimus albus*). Waterfowl also found in the lake include American coot (*Fulica americana*), American wigeon (*Anas americana*), gadwall (*Anas strepera*), and mallard (*Anas platyrhynchos*), eat many different species of aquatic vegetation. Nine species of turtles, as well as the federally and state threatened American alligator (*Alligator mississippiensis*), are found in the area. Furthermore, fifteen species of frogs and toads complete their amphibious lives in and around the waters of Lake Waccamaw.

Table 2: Significant wildlife species of Lake Waccamaw

Scientific Name	Common Name	Taxon	Federal Status	NC Status
Haliaeetus leucocephalus	Bald Eagle	Bird	BGEPA*	T
Alligator mississippiensis	American Alligator	Reptile	T(S/A)	T
Deirochelys reticularia	Chicken Turtle	Reptile	-	SR
Clemmys guttata	Spotted Turtle	Reptile	-	WL
Kinosternon baurii	Striped Mud Turtle	Reptile	-	WL

Native Vegetation

According to a survey of submersed aquatic vegetation (SAV) completed in November, 2012, by researchers at North Carolina State University, native aquatic vegetation is now found in 3,600 out of the roughly 9,000 acres of the open lake. The majority of the native SAV community is composed of southern naiad, slender pondweed (*Potamogeton pusillus*), the green alga *Nitella* (which is found in deeper parts of the lake), maidencane, and narrow-leaf spatterdock. Also found in this survey were floating heart (*Nymphoides aquatica*), a native primrose (*Ludwigia* sp.), slender naiad (*Najas minor*), and fanwort (*Cabomba caroliniana*). Additional species found along the northern shoreline in recent years include eel grass (*Vallisneria americana)*, American lotus (*Nelumbo lutea*), and longleaf pondweed (*Potamogeton nodosus*) (Diane Lauritsen, personal communication).

Along the undisturbed southern shore, aquatic vegetation comprises Natural Lakeshore community types unique to Lake Waccamaw, including the state endangered northeastern bladderwort (*Utricularia resupinata*) and globe-fruited seedbox (*Ludwigia sphaerocarpa*). Maidencane beds provide cover for fish, including the endemic Waccamaw darter and Waccamaw killifish. Pond cypress and bald cypress trees rim the lakeshore, having stood the test of time and weather for hundreds of years. The Venus hair fern, (*Adiantum capillus-veneris*), listed as threatened in NC, is found on the limestone bluff along the northern shore of the lake.

Table 3: Significant vascular plants of Lake Waccamaw

Scientific Name	Common Name	Federal Status	NC Status
Ludwigia sphaerocarpa	Globe-fruit Seedbox	-	E
Utricularia resupinata	Northeastern Bladderwort	-	E
Adiantum capillus-veneris	Venus Hair Fern	-	T
Bacopa caroliniana	Blue Water-hyssop	-	T
Epidendrum magnoliae	Green Fly Orchid	-	T
Sagittaria isoetiformis	Quillwort Arrowhead	-	T
Utricularia cornuta	Horned Bladderwort	-	T
Dionaea muscipula	Venus Flytrap	FSC	SC-V
Eriocaulon aquaticum	Seven-angled Pipewort	-	SC-V
Spiranthes laciniata	Lace-lip Ladies'-tresses	-	SC-V
Boltonia asteroides var. glastifolia	White Doll's-daisy	-	SR
Cladium mariscoides	Twig-rush	-	SR
Ludwigia brevipes	Long Beach Seedbox	-	SR

Luziola fluitans	Southern Water Grass	-	SR
Lycopus angustifolius	Southern Bog Water-horehound	-	SR
Sagittaria filiformis	Water Arrowhead	-	SR
Sclerolepis uniflora	One-flower Hardscale	-	SR
Cleistesiopsis divaricata	Spreading Pogonia	-	WL
Dichanthelium erectifolium	Erectleaf Witch Grass	-	WL
Habenaria repens	Water-spider Orchid	-	WL
Ilex cassine var. cassine	Dahoon	-	WL
Nelumbo lutea	American Lotus	-	WL
Nuphar sagittifolia	Narrowleaf Cowlily	-	WL
Panicum tenerum	Southeastern Panic Grass	-	WL
Rhexia cubensis	West Indies Meadow-beauty	-	WL
Rhynchospora nitens	Shortbeak Baldsedge	-	WL
Sarracenia flava	Yellow Pitcher-plant	-	WL
Sideroxylon lycioides	Buckthorn Bumelia	-	WL
Xyris smalliana	Small's Yellow-eyed Grass	-	WL

E. Predicted Environmental Effects of Project

1. Topography

Topography is a geologic feature, and as such, no impact is to be expected.

2. Soils

Studies have shown that fluridone residue in natural pond hydrosoil generally declines to non-detectable levels 16-52 weeks following an application. Dissipation of fluridone from the hydrosoil occurs by gradual adsorption into the water, where it is degraded primarily by photolysis. The products resulting from the photodegradation process do not persist or accumulate in the hydrosoil.

Endothall breaks down via microbial action. This biological process occurs at different rates depending on water temperature. Active compounds can persist for up to a week in cool water, or as little as 24 hours during the late summer. No direct, indirect, or cumulative impacts to soils are anticipated.

3. Land Use

N/A

4. Wetlands

Herbicides can impact submersed vegetation in wetland communities. However, the application of granular herbicide, at low concentrations, will greatly reduce the likelihood of impacting non-target wetlands via dispersion.

5. Prime or Unique Agricultural Lands

N/A

6. Public Lands and Scenic, Recreational, and State Natural Areas

Treating hydrilla with herbicides will not affect the designations currently assigned to Lake Waccamaw, including State Lake and State Park. The project's goal of eradicating hydrilla from Lake Waccamaw is consistent with the goal of establishing and promoting the Waccamaw River as a National Water Trail for its recreational opportunities and a National Blueway for a comprehensive watershed approach to resource stewardship. Failure to manage hydrilla in Lake Waccamaw will negatively impact these efforts, the lake's recreational potential and the long-term stewardship of its important resources.

No direct, indirect, or cumulative impacts to public lands and scenic, recreational, and State Natural Areas resulting from the proposed project are anticipated.

7. Areas of Archaeological or Historical Value

There should be no environmental impacts to these areas based on any of the treatment options.

8. Air Quality

The on-the-lake operations needed for this project will be conducted using motor boats and airboats (a.k.a. fanboats). These vessels are outfitted with petroleum-powered internal-combustion engines, and therefore release carbon monoxide, carbon dioxide and other emissions known to impact air quality. The operation of airboats is not anticipated to affect air quality any different than current vehicle traffic around the lake. The airboats are equipped with 4-stroke engines which produce comparatively less "greenhouse gas" than the typical 2-stroke boat motor. The quantity of petroleum used on the lake (by vessels and equipment) for this project will be outweighed by petroleum used on the road (by vehicles to transport personnel to and from the lake).

The herbicides and adjuvants used for this project will not affect air quality. No significant direct or indirect impact to air quality is expected.

9. Noise Levels

Some of the on-the-lake operations will be performed with the use of airboats. Airboats are generally louder than most motor boat engines and require the operators to wear hearing protection. The sound of an airboat is equivalent to a small prop-driven airplane. The elevated noise level can become a disruption to lake residents and recreationists, but this will be temporary and be limited to weekdays during daylight hours.

Elevated noise levels from airboats may also be temporarily disruptive to wildlife; however, lasting effects are not expected.

10. Water Resources

Per US Environmental Protection Agency labeling (SePRO Inc. website, http://www.sepro.com/documents/SonarONE_Label.pdf), there are no restrictions on use of fluridone-treated water for swimming, fishing, or consumption by domesticated animals (i.e., pets, cattle, etc). Similarly, endothall is deemed as a food-safe herbicide and does not impose restrictions on recreation activities following application. See the United Phosphorus Inc. website, http://www.upi-usa.com/aquatics/MSDS_Labels.php, for product labels and MSDS documents or appendix E for Aquathol-K label.

No direct, indirect, or cumulative impacts to water resources are anticipated.

11. Forest Resources

As this is a fully aquatic application of herbicides, there is no predicted impact to forest resources in the area.

12. Shellfish or Fish and Their Habitats

There are a number of rare and endemic fish, mollusks, and crustaceans in Lake Waccamaw that would be exposed to the herbicides and impacts to these populations are of particular concern. Fluridone is the main herbicide proposed for the management of hydrilla in the lake, with a target concentration ~10 parts per billion (ppb) maintained for up to 4 months. A report submitted to the U.S. Forest Service details the ecological risk of using fluridone (Durkin 2008). The report concludes that using the maximum target application rate of 150 parts per billion (as either acute or chronic exposure) does not result in any direct toxic effects in fish. The report further concludes that expected concentrations of fluridone in water would not have direct adverse effects on aquatic invertebrates. The acute and chronic toxicity values for sensitive and tolerant groups of aquatic invertebrates were only slightly greater than those for fish (Durkin 2008).

Endothall (Aquathol®), a contact herbicide, may also be used to treat hydrilla in several areas of Lake Waccamaw. This herbicide will not affect aquatic biota acutely or chronically when applied at concentrations recommended on the label (Compliance Services International 2001).

North Carolina State University has recently conducted acute toxicity tests using fluridone and endothall on glochidia (larval stage) and juvenile freshwater mussels. Toxicity testing was conducted on *Lampsilis siliquoidea*, which is in the same genus as four other mussel species in Lake Waccamaw (see Table 1). The results indicate that the concentrations of fluridone and endothall used for the treatment of hydrilla (5 parts per billion and 3 parts per million, respectively) are not toxic to the most sensitive life stages of freshwater mussels (Dr. Greg Cope, personal communication). For a full summary of results, please see Appendix C.

The decomposition of vegetation following herbicide treatment can lead to localized zones of low dissolved oxygen concentrations. This phenomenon can pose a threat to aquatic life.

The herbicides proposed in this treatment are not known to persist in the environment; the compounds break down rapidly via photolysis, hydrolysis, and microbial action. The chemicals are also not known to bio-accumulate.

No significant direct, indirect, or cumulative impacts to shellfish or fish and their habitats are anticipated.

13. Wildlife and Natural Vegetation

Proposed hydrilla management activities on Lake Waccamaw will not significantly affect wildlife. The successful management of hydrilla will make it unavailable for potential summer-fall use by waterfowl and other species that could utilize surface-matted dense plant biomass. However, there is an abundance of native aquatic vegetation, the usual food source for this wildlife.

In addition to delimiting hydrilla, NC State University inventoried the following aquatic plants during lake-wide aquatic vegetation surveys (conducted in November 2012):

- Southern Naiad*
- Slender Pondweed (*Potamogeton pusillus var. pusillus*)*
- Nitella (*Nitella* sp.)*
- Maidencane*
- Narrowleaf Spatterdock*
- Big Floating Heart (*Nymphoides aquatica*)
- Primrose (*Ludwigia* sp.)
- Slender Naiad (*Najas gracillima*)
- Cabomba (*Cabomba caroliniana*)

* Member of the species assemblage which accounts for >98% of all native aquatic vegetation in the lake.

The two submersed naiad species have similar sensitivity to fluridone (relative to hydrilla). In the NW treatment area (950+ acres in red below), these species will be removed or strongly suppressed. However, most native submersed plant biomass is found in other areas of the lake (see SAV biovolume map below), so the overall lake-wide impact to these species will be minimal due to strong dilution of the herbicide away from the treated zone. Spatterdock, floating heart, cabomba, and the pondweed species are all approximately two times less susceptible than hydrilla and native naiad species. In treated areas, these plants may show some temporary chlorosis and growth suppression during the active treatment period but no long-term reduction in abundance. At proposed use rates, Nitella, maidencane, and primrose are tolerant of treatment. In total, especially when balanced against the imminent risk of lake-wide plant community changes from an unmanaged hydrilla infestation, the proposed treatment program has an acceptable low risk to the native aquatic vegetation in Lake Waccamaw. A two-acre area in the southwest quadrant of the lake may be managed using herbicide(s) other than fluridone. The dominant native plant in this area is the emersed maidencane. This community of maidencaine should not be impacted by endothall use but may be incur some injury depending on treatment strategy. Several rare and uncommon species can be found in close proximity to the treatment area and dam, including the state threatened quillwort arrowhead (*Sagittaria isoetiformis*) and the significantly rare southern water grass (*Luziola fluitans*). *Sagittaria* species have shown low susceptibility to both fluridone and endothall (M. Heilman, personal communication), but southern water grass has shown sensitivity to fluridone.

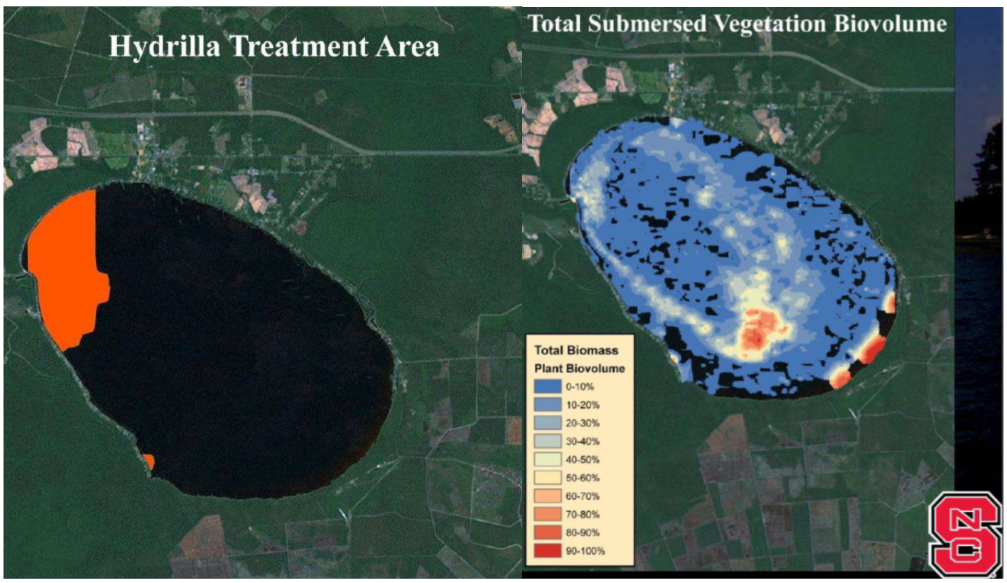


Figure 2. Proposed hydrilla treatment areas (left image) and total submersed plant biovolume (% of watercolumn with submersed vegetation) with blue (low) to red (high) color coding (right image).

14. Introduction of Toxic Substances

All persons conducting work on the lake related to this project will insure that toxic substances (e.g., fuel, oil, lubricants, herbicides, adjuvants, etc.) as well as their containers will be handled in accordance with all appropriate State and Federal regulations. There will be no littering and products/containers will be recycled whenever feasible. No significant direct, indirect, or cumulative impact to the environment is expected as a result of the use these substances.

F. Mitigative Measures

1. Topography

Not applicable.

2. Soils

Not applicable.

3. Land Use

Not applicable.

4. Wetlands

Efforts will be made to minimize the exposure of natural wetland communities, such as the Natural Lake Marsh Shoreline Swamp (Lake Waccamaw Subtype) and the Natural Lake Shoreline Marsh (Lake Waccamaw Pond-Lily Subtype) to herbicide. Vegetation will be sampled periodically at existing plots, using protocols from the Carolina Vegetation Surveys (Peet et al., 1998), to monitor condition. Additional plots will be supplemented if needed.

If wetland communities are shown to be affected, then herbicide prescriptions will be re-evaluated and altered as needed.

5. Prime or Unique Agricultural Lands

Not applicable.

6. Public Lands and Scenic, Recreational, and State Natural Areas

Not applicable.

7. Areas of Archaeological or Historical Value

Not applicable.

8. Air Quality

Not applicable.

9. Noise Levels

Not applicable.

10. Water Resources

Physical and chemical characteristics of water quality are taken twice monthly (2nd and 4th Wednesday mornings of each month) at four locations on or adjacent to Lake Waccamaw by trained volunteers as part of the Waccamaw River Volunteer Monitoring Project administered by the Waccamaw Riverkeeper, Lake Waccamaw State Park and Coastal Carolina University's Waccamaw Watershed Academy. Locations for volunteer water quality monitoring are: Big Creek, a major tributary to the Lake; Maple Street, in the Lake proper; Canal Cove, in the canal prior to its discharge into the Lake; and Dam, in the Lake proper by the check dam. Water quality parameters measured are: Dissolved Oxygen, Temperature, pH, Conductivity, Total Dissolved Solids, Nutrients (Nitrate, Nitrite and Ammonia), Turbidity and Bacteria (Fecal Coliform and E. Coli). These data are published following an intensive quality assurance/quality control procedure at:
http://bccmws.coastal.edu/volunteermonitoring/data_access.php?proj_id=1.
More information on the program is available at:
http://www.coastal.edu/wwa/vm/index.html.

11. Forest Resources

Not applicable.

12. Shellfish or Fish and Their Habitats

The decomposition of vegetation following herbicide treatment can drive dissolved oxygen levels down to a point that impacts aquatic organisms. This can be mitigated or avoided by coordinating correct timing of the herbicide applications. Applying herbicide when hydrilla is just beginning to grow (late spring) is advantageous since there is less biomass to decompose. Cooler spring-time water contains greater amounts of dissolved oxygen than summer-time water temperatures. Additionally, plants respond slowly to fluridone, unlike contact herbicides that lead to more abrupt death and decomposition. The potential for a dissolved oxygen related fish kill is more of a concern in heavily vegetated systems that are subjected to whole pond/lake treatments.

North Carolina Wildlife Resources Commission and North Carolina State Parks will continue to conduct standardized surveys for priority species of fish and mollusks in Lake Waccamaw. Species density and/or relative abundance data (catch per unit effort) has been and will continue to be compared among sampling years to determine the status of fish and mollusk populations and if management activities need modification.

Methods

Fish surveys

In 2009, WRC and DPR established three long-term monitoring stations, located on the southwest shore of the lake near the dam, the north shore near Flemington Drive, and the southeast shore near the mouth of Big Creek. A fourth survey location will be added in the treatment area and will be sampled prior to the application of the herbicide. Fish will be sampled using 3.1 and 4.6 m seines. The number of seine hauls and duration of sampling will be recorded to determine catch per unit effort (CPUE). In addition, biologists will conduct underwater visual surveys for broadtail madtoms (*Noturus* sp. 2) and Waccamaw darters which may not be easily collected using seines or dip nets. CPUE is determined by collection technique (seine or visual) for each site. CPUE is expressed as the number of individual fish collected per minute of seining (indiv/min), or the number of fish observed per person-minute (indiv/p-min). Twenty-five individuals of each priority fish species will be measured (total length) to assess the size distribution and released. Fish collections will be scheduled on an annual basis during the first seven years of treatment, at which time the need for future monitoring will be evaluated.

Mollusk surveys

Using the location of previous mussel surveys and quantitative pilot surveys, WRC and DPR biologists established two long-term monitoring stations for mussels and non-hydrobiid snails in Lake Waccamaw in 2009. The first site is located on the south shore of the lake near the Lake Waccamaw State Park pier and the second site is located on the north shore of the lake near Flemington Drive. A third quantitative mussel sampling location will be added inside the treatment area and surveyed prior to the application of the herbicide.

Mussels and snails are collected using a systematic sampling design with random starts (Strayer and Smith 2003; Pooler and Smith 2005) at each 30m X 30m site. Quadrats (0.25 m^2) are sampled at regular distances from a random starting point. The sites are divided into a grid; three random starting points are generated from a random numbers table. Each quadrat that follows a random start is part of the systematic (or sampling) unit. Each random start consists of 36 quadrats for a total of 108 quadrats per site. The interval between quadrats is 5 m. Surveyors collect mussels from the substrate surface, and burrowed individuals are also collected at a subset of quadrats (50%) by excavating the substrate to 10 cm. Excavating a portion of the quadrats produces a better estimate of abundance and does not greatly increase sampling effort (Strayer and Smith 2003). Mussels found on the surface are recorded separately from those that are burrowed. Fifty individuals of each mussel species are measured to assess the size distribution. Mussels are returned to the study site after identification and measurement. Data is analyzed using the USGS Mussel Estimation Program Version 1.5.2 (USGS Leetown

Science Center, Aquatic Ecology Lab). This program uses the total counts from the excavated quadrates to calibrate the abundance of the surface counts and will compute population density and abundance estimates. Mollusks will be sampled every other year during the first 7 years of treatment, at which time the need for future monitoring will be evaluated.

Habitat conditions are recorded at each sampling location. These included substrate composition, water depth (m), water clarity (horizontal Secchi distance in m), temperature (ºC), dissolved oxygen (mg/l), conductivity (µs), and pH. Digital photographs of fish and mussel specimens and sampling locations are taken. Voucher specimens are preserved when necessary and deposited in the NC Museum of Natural Sciences.

Hydrobiid snails will be sampled on an annual basis for the first seven years of treatment, at which time the need for future monitoring will be evaluated. Four sites have been identified: one at each of the two previously established mollusk sampling areas, plus one at each newly identified hydrilla management area. Each site will be sampled with a Petite Ponar dredge (sample area 0.023 m^2), with 11 replicates per site. The substrate will be sieved through a mesh screen, and snails will be sorted into sample jars for later identification. All vouchers will be deposited with the NC State Museum of Natural Sciences.

13. Wildlife and Natural Vegetation

Wildlife

Not applicable.

Native Vegetation

Efforts will be made to adaptively mitigate stress on native vegetation while accomplishing the primary hydrilla control objective. The primary mitigative strategy is to conduct partial-lake treatments which will minimize herbicide exposure to aquatic plants in most areas of the lake. The second mitigative strategy is to implement the use of narrow spectrum herbicides (Getsinger et al. 2011). Hydrilla is particularly sensitive to fluridone. Targeting a low-concentration of fluridone should reduce impacts to native aquatic vegetation in the lake. The third mitigative strategy will be to implement an early detection rapid response protocol for dealing with any colonies of hydrilla that develop outside of the infestation epicenter. During the course of this project SAV surveys will be ongoing as an effort to detect "satellite" hydrilla beds and monitor the assembly and condition of SAV lake-wide. Responding early to "satellite" hydrilla beds will reduce the total acreage of lake that needs treatment, and therefore minimize the total amount of herbicide product applied. Delayed treatments or slow responses to hydrilla only allow the plants to spread further and increase total acreage that would then require management.

Surveys of submersed aquatic vegetation will be conducted using a point intercept sampling technique. Guided by predetermined points on a grid with specific GPS locations the surveyor(s) will sample each point using an aquatic weed rake. Recovered plants will be identified to species and recorded. Any specimens that cannot be identified to species in the field will be sent to and vouchered by North Carolina State University's herbarium. Approximately 400 individual points will be sampled at each sampling event. If aquatic vegetation surveys indicate a decline in native submerged aquatic vegetation outside of the treatment areas is occurring then the herbicide prescriptions will be re-evaluated.

Several species of plants may be impacted by herbicide flow and dissipation from the treatment area near the dam, including the state threatened globe-fruited seedbox and southern water grass. Pre-application surveys will record the extent of these plants and post-application surveys will determine if herbicides are impacting those communities. If significant impacts to these plants are occurring than possible mitigative and corrective actions include:

1) Alter herbicide selection and/or application timing.
2) Remove a subset of the population, hold and propagate at North Carolina State University (or other off-site location), and reintroduce upon project completion.
3) Reintroduce individuals to impacted areas that are collected from wild populations.

Note: A potential mitigative measure would involve lowering the water level in the lake so that less herbicide product is required. This goes back to the concepts of targeting a concentration of herbicide in the water and maintaining exposure time (CET) which is described on page 3. Currently, the water level at Lake Waccamaw is stabilized with the dam structure. One option that may be explored is to repair the water-level control device on the dam or modify the dam so that the water level in the lake can be lowered.

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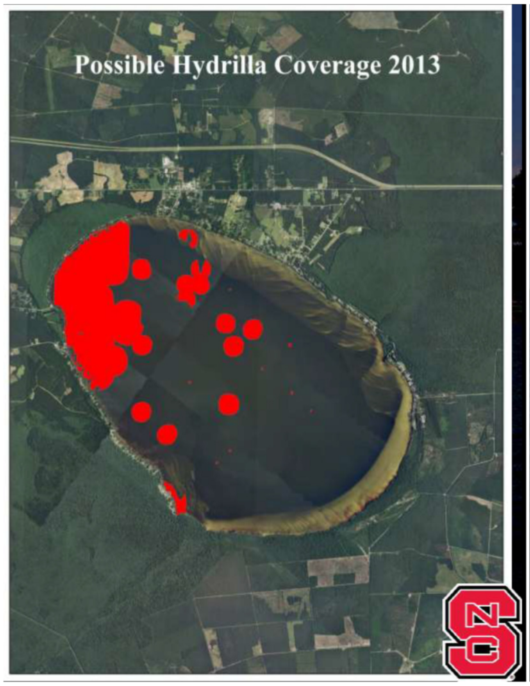
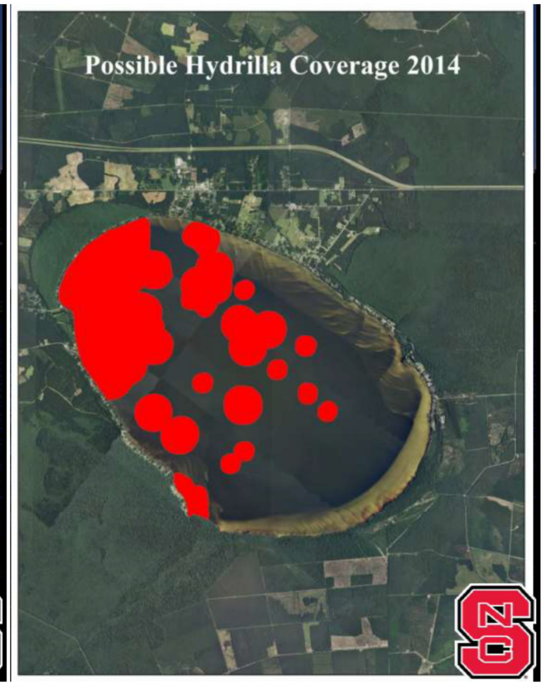
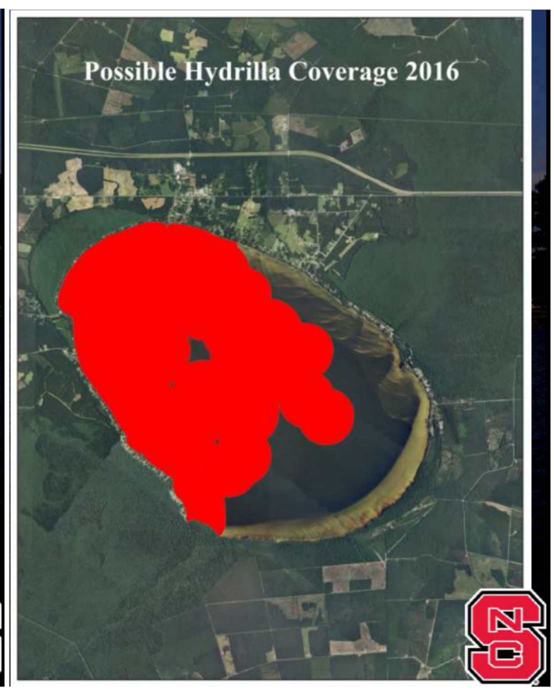
H. Exhibits

EXHIBIT 1: Progression Model of Hydrilla in Lake Waccamaw if left untreated (2013-2017)





Predicted Coverage of Lake in 2013 = 1474 acres (left image); 2014 = 2932 acres (right image)



Predicted Coverage of Lake in 2015 = 4596 acres (left image); 2016 = 5700 acres (right image)

Predicted Coverage of Lake in 2017 = 6223 acres

I. State and Federal Permits Required

1. State of North Carolina, Department of Environment and Natural Resources, Division of Water Quality General Permit NCG50000 to discharge pesticides under the National Pollutant Discharge Elimination System.

2. North Carolina Environmental Policy Act (As Amended).
   The EA has been developed in accordance with the requirements of the State Clearinghouse review process under the North Carolina Environmental Policy Act (NCEPA, G.S. 113A-1), based upon an agreement with the North Carolina Department of Environment and Natural Resources.

3. Endangered Species Act of 1973.
   Coordination with the U.S. Fish and Wildlife Service includes consultation under Section 7 of the Endangered Species Act of 1973, as amended.

4. Depending on the involvement of certain funding sources a NEPA document may be drafted for this project.

Appendix A: Lake Waccamaw Hydrilla Management Technical Advisory Committee.

NC Department of Agriculture and Consumer Services
- Rick Iverson, Weed Specialist – Plant Industry Division

NC Division of Parks and Recreation
- Toby Hall, Superintendent – Lake Waccamaw State Park
- Jonathan Short, Ranger – Lake Waccamaw State Park
- Kristen Woodruff, Superintendent – Lake Waccamaw State Park
- Angelia Allcox, District Superintendent
- Jon Blanchard, Program Head – Natural Resources Program
- Jamie Sasser, Coastal Region Biologist – Natural Resources Program
- Ed Corey, Inventory Biologist – Natural Resources Program

NC Division of Water Quality
- Stephanie Garret, Senior Environmental Technician – Surface Water Protection
- Mark Vander Borgh, Environmental Biologist

NC Division of Water Resources
- Rob Emens, Invasive Species Specialist

NC Natural Heritage Program
- Laura Gadd, Botanist
- Judith Ratcliffe, Aquatic Ecologist
- Mike Schafale, Community Ecologist

NC Wildlife Resources Commission
- Keith Ashley, District 4 Fisheries Biologist - Division of Inland Fisheries
- Ryan Heise, Aquatic Wildlife Diversity Program Research Coordinator

US Fish and Wildlife Service
- Sarah McRae, Aquatic Endangered Species Biologist

North Carolina State University
- Greg Cope, Professor of Toxicology
- Brett Hartis, Aquatics Extension Associate, Crop Science Department
- Steve Hoyle, Research Specialist, Crop Science Department
- Justin Nawrocki, Graduate Research Assistant, Crop Science Department
- Rob Richardson, Associate Professor and Extension Specialist, Crop Science Department

University of North Carolina – Wilmington
- Larry Cahoon, Professor of Biology and Marine Biology

The Nature Conservancy
- Dan Ryan, Program Director – Southeast Coastal Plain Program

Winyah Rivers Foundation
- Christine Ellis, Waccamaw RIVERKEEPER ®

Invasive Plant Control, Inc.
- Randy Westbrooks, Invasive Species Prevention Specialist

SePRO Corporation
- Mark Heilman, Aquatics Technology Leader
- Sarah Miller, Aquatics Specialist

Other Members
- Diane Lauritsen
- Steve Smith

Appendix B: State and Federal Status Definitions.

Federal:
T – Threatened – A taxon which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range.
(Endangered Species Act, Section 3).

T(S/A) – Threatened due to similarity of appearance – Section 4 (e) of the [Endangered Species] Act authorizes the treatment of a species (subspecies or population segment) as endangered or threatened even though it is not otherwise listed as endangered or threatened if -- (a) the species so closely resembles in appearance an endangered or threatened species that enforcement personnel would have substantial difficulty in differentiating between the listed and unlisted species; (b) the effect of this substantial difficulty is an additional threat to an endangered or threatened species; and (c) such treatment of an unlisted species will substantially facilitate the enforcement and further the policy of the Act.
(*Federal Register*, November 4, 1997)

BGEPA – Bald and Golden Eagle Protection Act – The Bald and Golden Eagle Protection Act (16 U.S.C. 668-668c), enacted in 1940, and amended several times since then, prohibits anyone, without a permit issued by the Secretary of the Interior, from "taking" bald eagles, including their parts, nests, or eggs. The Act provides criminal penalties for persons who "take, possess, sell, purchase, barter, offer to sell, purchase or barter, transport, export or import, at any time or any manner, any bald eagle ... [or any golden eagle], alive or dead, or any part, nest, or egg thereof." The Act defines "take" as "pursue, shoot, shoot at, poison, wound, kill, capture, trap, collect, molest or disturb."

- For purposes of these guidelines, "disturb" means: “to agitate or bother a bald or golden eagle to a degree that causes, or is likely to cause, based on the best scientific information available, 1) injury to an eagle, 2) a decrease in its productivity, by substantially interfering with normal breeding, feeding, or sheltering behavior, or 3) nest abandonment, by substantially interfering with normal breeding, feeding, or sheltering behavior."

- In addition to immediate impacts, this definition also covers impacts that result from human-induced alterations initiated around a previously used nest site during a time when eagles are not present, if, upon the eagle's return, such alterations agitate or bother an eagle to a degree that interferes with or interrupts normal breeding, feeding, or sheltering habits, and causes injury, death or nest abandonment.

- A violation of the Act can result in a fine of $100,000 ($200,000 for organizations), imprisonment for one year, or both, for a first offense. Penalties increase substantially for additional offenses, and a second violation of this Act is a felony. (http://www.fws.gov/midwest/eagle/guidelines/bgepa.html)

FSC – Federal Species of Concern – A species under consideration for listing, for which there is insufficient information to support listing at this time. These species may or may not be listed in the future (Gadd and Finnegan, 2012).

State:

<u>Animals</u>
E – Endangered - Any native or once-native species of wild animal whose continued existence as a viable component of the State's fauna is determined by the Wildlife Resources Commission to be in jeopardy or any species of wild animal determined to be an 'endangered species' pursuant to the Endangered Species Act. (Article 25 of Chapter 113 of the General Statutes: 1987)

T – Threatened - Any native or once-native species of wild animal which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range, or one that is designated as a threatened species pursuant to the Endangered Species Act. (Article 25 of Chapter 113 of the General Statutes: 1987)

SC – Special Concern - Any species of wild animal native or once-native to North Carolina which is determined by the Wildlife Resources Commission to require monitoring but which may be taken under regulations adopted under the provisions of this Article. (Article 25 of Chapter 113 of the General Statutes: 1987)

SR – Significantly Rare – "Any species which has not been listed by the N.C. Wildlife Resources Commission as an Endangered, Threatened, or Special Concern species, but which exists in the state (or recently occurred in the state) in small numbers and has been determined by the N.C. Natural Heritage Program to need monitoring." (LeGrand et. al, 2012)

WL – Watch List – "Any other species believed to be rare and of conservation concern in the state but not warranting active monitoring at this time." (LeGrand et. al, 2012)

<u>Plants</u>
E – Endangered - Any species or higher taxon of plant whose continued existence as a viable component of the State's flora is determined to be in jeopardy. (General Statutes 19B 106:202.12)

T – Threatened - Any resident species of plant which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range. (General Statutes 19B 106:202.12

SC-V – Any species or higher taxon of plant which is likely to become a threatened species within the foreseeable future. (NCAC 02 NCAC 48F.0401)

SR – Significantly Rare - Any species not listed by the N.C. Plant Conservation Program as Endangered, Threatened, or Candidate, which is rare in North Carolina, generally with 1-100 populations in the state, frequently substantially reduced in numbers by habitat destruction (and sometimes also by direct exploitation or disease). (Gadd and Finnegan, 2012)

WL – Watch List - Any other species believed to be rare and of conservation concern in the state but not warranting active monitoring at this time. (Gadd and Finnegan, 2012)

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Appendix C: Summary of Acute Toxicity of the Aquatic Herbicides Sonar PR® (fluridone) and Aquathol-K® (endothall) on the Early Life Stages of the Unionid Mussel, *Lampsilis siliquoidea*.

Jennifer M. Archambault[1], Christine M. Bergeron[2], Angela G. White[1], and W. Gregory Cope[2]

[1] North Carolina Cooperative Fish and Wildlife Research Unit, Department of Biology, Box 7617, North Carolina State University, Raleigh, North Carolina 27695 USA;

[2] Department of Environmental and Molecular Toxicology, Box 7633, North Carolina State University, Raleigh, North Carolina 27695 USA.

Toxicological information was generated on two aquatic herbicides commonly used to treat hydrilla (*Hydrilla verticillata*), an invasive aquatic weed that was detected recently at Lake Waccamaw in North Carolina. The following is a brief summary of findings from experiments evaluating the acute toxicity of formulation-grade Sonar PR® (fluridone) and Aquathol-K® (endothall) on the early life stages of a native freshwater mussel belonging to the family Unionidae.

Methods

Test Organisms

The freshwater mussel *Lampsilis siliquoidea* (fatmucket) was used for all toxicity tests; *L. siliquoidea* is routinely used in toxicity testing due to its wide availability and ease of laboratory culture. This species was selected for this study due to the urgent timeline for gathering information on the herbicides proposed for use at Lake Waccamaw and because the larval and juvenile life stages were immediately available. *Lampsilis siliquoidea* is a congener of the Waccamaw fatmucket (*Lampsilis fullerkati*), one of the imperiled resident species of Lake Waccamaw. All mussels were supplied by the mussel culture laboratory at Missouri State University (Springfield, Missouri), and all juveniles were propagated via host-fish infection, using standard propagation and culture methods (Barnhart 2006).

Glochidia assessment

Glochidia were < 24 h old at the start of each test, and they were acclimated to reconstituted soft water and the test temperature of 20°C for 2 h before the beginning of the experiments (ASTM 2006a, 2006b). Reconstituted soft water was selected for use due to similarity of water chemistry (i.e., alkalinity and hardness) with Lake Waccamaw (Stager and Cahoon 1987). Tests were 48-h nonaerated static experiments conducted according to the American Society of Testing and Materials guidelines for glochidia (ASTM 2006a). Survival was assessed at 24 h for a subsample of approximately 50 of the 150 glochidia in each of three replicates per treatment. A saturated sodium chloride solution was used to stimulate a shell-closure response; glochidia that were closed before adding the saturated sodium chloride and those that did not respond to the salt addition were considered nonviable. Salt closure responses were documented with either an Olympus SZ61 microscope (Olympus America, Center Valley, Pennsylvania) and QCapture Pro 5.1 digital photographic software (Quantitative Imaging Corporation, Burnaby, British Columbia, Canada) or a Leica EZ4 D stereo microscope with integral digital camera and Leica Application Suite EZ digital photographic software (Leica Microsystems, Ltd., Switzerland).

Juvenile assessment

Juveniles were acclimated to reconstituted soft water and the test temperature of 20°C for 2 days before the start of experiments. Experiments were 96-h nonaerated static-renewal tests with 90% reconstituted soft water and chemical renewed at 48 h (ASTM 2006a, 2006b). Survival was assessed visually at 48 and 96 h with an Olympus SZ61 microscope to detect foot movement outside of the shell, foot movement inside the shell, or the presence of a heartbeat for the seven mussels in each of three replicates per treatment. Controls included 10 mussels per replicate. Juveniles were 3 – 5 days old at the start of each test and averaged 252 µm (± 14 µm).

Test Conditions

Herbicide treatment concentrations were informed by recommended application rates for treatment of hydrilla, label application rates, and acute toxicity data reported on Material Data Safety Sheets (MSDS). A stock solution of formulation-grade Sonar PR® (fluridone) was prepared at 1.304 mg/L and provided by SePRO Research and Technology Campus (Whitakers, North Carolina). Test concentrations of Sonar PR® ranged from 2.5 to 200 µg/L, with an additional treatment at the stock solution concentration (1,304 µg/L). A concentrated formulation-grade sample of Aquathol-K® (endothall), labeled as 4.23 lb/gal (506,865.79 mg/L), was provided by the Crop Science Department, North Carolina State University (Raleigh, North Carolina) and subsequently diluted to a working stock of 1,000 mg/L. Test concentrations of Aquathol-K® ranged from 0.5 to 1,000 mg/L. Quality assurance and control were ensured by conducting all tests according to the Standard Guide for Conducting Laboratory Toxicity Tests with Freshwater Mussels (ASTM 2006a). Tests were conducted in light- and temperature-controlled environmental chambers (Precision Model 818, Thermo Fisher Scientific, Marietta, Ohio, and Isotemp Model 146E, Fisher Scientific, Dubuque, Iowa). Mean water quality conditions among all tests were 27.8 mg $CaCO_3$/L alkalinity, 39.0 mg $CaCO_3$/L hardness, 219.6 µS/cm conductivity, 7.50 pH, and 8.47 mg/L dissolved oxygen (n = 4 for alkalinity and hardness, n = 36 for all other variables).

Statistical Analysis

Survival data from both glochidia and juvenile experiments were used to generate median effective concentrations (EC50) and 95% confidence intervals (CI) using the Trimmed Spearman-Karber method (Comprehensive Environmental Toxicity Information Software (CETIS) ™, v1.8.0.12, Tidepool Scientific, LLC, McKinleyville, California, USA). The EC50 was defined as the concentration that caused mortality in 50% of the individuals in the exposed sample, and the EC05 was defined as the concentration that caused mortality in 5% of the sample. EC50s were considered significantly different when their 95% CIs did not overlap.

Results and Discussion

Control viability at 24 and 48 h in glochidia tests were > 90% of initial viability on arrival to the laboratory for all experiments. Control survival in experiments with juveniles was > 90%, except at the 96-h time point in the Aquathol-K® experiment. While the control survival (73.3%) at 96 h in the Aquathol-K® experiment was below the 80% recommended in the standard guideline for toxicity tests with freshwater mussels (ASTM 2006a), results are reported herein because survival was > 90% in three of the lowest treatment concentrations at 96 h.

Concentration verification

Exposure accuracy (i.e., measured herbicide concentration compared to target concentration) was calculated as: exposure accuracy = $(P_m)/(P_t) \cdot 100$, where P_m is the measured herbicide concentration and P_t is the target concentration. The measured concentration of the fluridone stock solution provided by SePRO Research and Technology Campus was 108.3% of the reported concentration (1,304 µg/L), and the mean exposure accuracy in experiments with fluridone was 119.9% (range 102 – 176%) of target treatment concentrations. In experiments with Aquathol-K®, samples were collected for concentration verification and results are pending.

Sonar PR® toxicity

The glochidial EC50 at 24 h was 864.9 µg/L [parts per billion; (95% CI, 729.3 – 1,026 µg/L)], and the 48-h EC50 was 977.6 µg/L (787.4 – 1,214 µg/L). The Sonar PR® experiment with juveniles yielded a 48-h EC50 of 1,197 µg/L (568.6 – 2,522 µg/L) and a 96-h EC50 of 510.7 µg/L [309.4 – 842.9 µg/L; (Table 1)]. The 24-h EC05 for glochidia was 290.2 µg/L (0 – 597.6 µg/L); EC05s in the juvenile tests and at the 48-h time point of the glochidia test were not determined due to lack of partial mortality responses.

Table 1. Acute median effective concentrations (EC50s) for glochidia and juvenile *Lampsilis siliquoidea* exposed to herbicides proposed for use in treating *Hydrilla verticillata* at Lake Waccamaw State Park, North Carolina.

Life stage	Time point (h)	Sonar PR® (fluridone; µg/L)	Aquathol-K® (endothall; mg/L)
glochidia	24	864.9 (729.3 – 1026)	31.2 (30.3 – 32.2)
	48	977.6 (787.0 – 1214)	27.6 (25.5 – 29.9)
juveniles	48	1197 (568.6 – 2522)	214.1 (134.0 – 342.0)
	96	510.7 (309.4 – 842.9)	34.4 (29.3 – 40.5)

All of the EC50s generated for Sonar PR®, including those generally reported for regulatory purposes (24 h for glochidia, 96 h for juveniles), are two or more orders of magnitude greater than the water column target for Lake Waccamaw (5 µg/L) and are greater than three times more than the maximum label application rate of 150 µg/L. Our results indicate that the early life stages of *L. siliquoidea* are much more acutely sensitive to Sonar PR® than some other aquatic organisms; toxicity data available on the MSDS lists EC50s and LC50s for *Daphnia* (48-h) and fish (96-h; no species listed) that range from 3.6 to 4.5 parts per million.

Aquathol-K® toxicity

The glochidial EC50 at 24 h was 31.2 mg/L [parts per million; (30.3 – 32.2 mg/L)], and the 48-h EC50 was 27.6 mg/L (25.5 – 29.9 mg/L). The Aquathol-K® experiment with juveniles yielded a 48-h EC50 of 214.1 mg/L (134 – 342 mg/L) and a 96-h EC50 of 34.4 mg/L [29.3 – 40.5 mg/L; (Table 1)]. The 48-h EC05 for juveniles was 34.6 mg/L (3.9 – 80.0 mg/L); EC05s in the glochidia tests and at the 96-h time point of the glochidia test were not determined due to lack of partial mortality responses or poor fit.

The label application rate of Aquathol-K® for treatment of hydrilla is 1 – 5 mg/L. The 24-h glochidial EC50 and 96-h juvenile EC50 – standard acute toxicity values for freshwater mussels – are approximately one order of magnitude greater than the application rate, indicating a relatively smaller margin of error in applying the herbicide compared to the Sonar PR®.

There was good agreement among the EC50s for glochidia and the 96-h juvenile EC50. Our results indicate there may be a defined threshold of tolerance; most mussels survived at concentrations ≤ 10 mg/L and experienced near complete mortality at concentrations ≥ 100 mg/L. Interestingly, the juvenile 48-h EC05 is similar to the 96-h EC50; however, more research is needed to determine if the 48-h time point has any predictive power for the 96-h acute toxicity.

Our results indicate that freshwater mussels are more acutely sensitive to Aquathol-K® than many aquatic or terrestrial organisms. The MSDS reports EC50s for 11 species ranging from > 100 mg/L to 1,071 mg/L, with the channel catfish (*Ictalurus punctatus*) and coho salmon (*Oncorhynchus kisutch*) as the most sensitive species and the bluegill sunfish (*Lepomis* *macrochirus*) as the most tolerant. The nearest relative to unionid mussels included in the ecotoxicity data is the eastern oyster (*Crassostrea virginica*), which has a 96-h EC50 (shell deposition) of 335 mg/L, approximately 10 times greater than the glochidial and juvenile EC50s for *L. siliquoidea*.

In summary, the two aquatic herbicides tested do not pose a substantive acute toxic risk to early life stages of a unionid mussel at or near label recommended treatment concentrations.

Literature Cited

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ASTM (American Society for Testing and Materials). 2006b. Standard guide for conducting acute toxicity tests with fishes, macroinvertebrates, and amphibians. E729-88a. ASTM International, West Conshohocken, Pennsylvania.

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Appendix D: Aquatic Herbicide Manufacturers.

1. SePRO Corporation http://www.sepro.com/

   Products:
   Sonar One ® (Fluridone)
   Sonar PR ® (Fluridone)
   Sonar Q ® (Fluridone)

2. Syngenta Crop Protection, Inc. http://www.syngenta-us.com/home.aspx

   Products:
   Reward ® (Diquat)

3. United Phosphorus, Inc. http://www.upi-usa.com/

   Products:
   Aquathol-K ® (Endothall)

4. Valent http://www.valent.com/

   Products:
   Clipper ® (Flumioxazin)
   Tradewind ® (Bispyribac)

Lake Waccamaw Fish and Mollusk Status Surveys

2015 Progress Report

Ryan J. Heise

Funded by
State Wildlife Grant Program,
and
Section 6 Recovery Funds
and
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North Carolina Wildlife Resources Commission
Aquatic Wildlife Diversity Program
Raleigh, NC

2015

Lake Waccamaw Fish and Mollusk Status Surveys

2015 Progress Report

Lake Waccamaw is a 3,615 hectare Carolina bay lake located in Columbus County, North Carolina. This natural lake is unique because of its water chemistry and large size, as compared to other bay lakes. Normally bay lakes are highly acidic, but due to the limestone formation that underlies the lake, the pH of the water is near neutral. Big Creek is the largest tributary that flows into the lake and the Waccamaw River forms at the outfall located on southeast end of the lake (Figure 1).

There are 17 species of rare fish and mollusks that have been documented from Lake Waccamaw (Table 1; Lindquist and Yarbrough 1979, Porter 1985, Mottesi 1998, Shute et al. 2000). Three species of fish are known only from the lake and the Waccamaw River directly below the lake. The Waccamaw Silverside (*Menidia extensa*) is listed by the US Fish & Wildlife Service (USFWS) as threatened and the entire lake and a short reach of lower Big Creek (USFWS 1993) are designated as critical habitat. Other endemic fishes include the Waccamaw Darter (*Etheostoma perlongum*), and the Waccamaw Killifish (*Fundulus waccamensis*). The Broadtail Madtom (*Noturus sp.*) is another rare species that has been found in the lake and nearby counties.

The endemic species of mollusks in Lake Waccamaw include the Waccamaw Fatmucket (*Lampsilis fullerkati*), Waccamaw Snail (*Amnicola sp. 1*), and the Waccamaw Siltsnail (*Floridobia sp.*). Other rare mussels located in Lake Waccamaw and elsewhere include the Pod Lance (*Elliptio folliculata*), Waccamaw Spike (*Elliptio waccamawensis*), Yellow Lampmussel (*Lampsilis cariosa*), Eastern Lampmussel (*Lampsilis radiata*), Rayed Pink Fatmucket (*Lampsilis splendida*), Tidewater Mucket (*Leptodea ochracea*), Savannah Lilliput (*Toxolasma pullus*), and

Eastern Creekshell (*Villosa delumbis*). Additional rare species of snails that have been collected in Lake Waccamaw include the Blackwater Ancylid (*Ferrissia hendersoni*) and Ridged Lioplax (*Lioplax subcarinata*).

One of the major goals of the North Carolina Wildlife Action Plan is to improve our understanding of species diversity in North Carolina, and to enhance our ability to make conservation decisions (NCWRC 2015). As part of this goal, we are conduct long-term monitoring surveys in areas that contain numerous endemic species, such as Lake Waccamaw. The results of this project support the USFWS recovery objectives for the Waccamaw Silverside by implementing a monitoring program (Part 6.0, USFWS 1993) to assess the status of the species and habitat quality of Lake Waccamaw.

Hydrilla (*Hydrilla verticillata*), a federal and state listed noxious weed, was first reported in Lake Waccamaw in October of 2012. This invasive plant poses a significant threat to the endemic species of Lake Waccamaw, especially Waccamaw Silversides, which prefer open water. The Lake Waccamaw Technical Advisory Committee (TAC), which includes members from universities, state and federal agencies, non-governmental organizations (NGOs), and other stakeholders, has developed a plan to control and ideally, eradicate Hydrilla, which includes the application of an herbicide (Fluridone). Due to the introduction of Hydrilla to Lake Waccamaw and the initiation of multiyear chemical treatment, we added an additional fish survey location within the infested area. Treatment of the Hydrilla began in June 2013 and has continued through the fall of 2015.

Black Mat Algae (*Lyngbya wollei*), a species that can become invasive, was collected by North Carolina State University in 2013 within the area already infested with Hydrilla. Surveys documented about 32 acres in 2013, 20 acres in 2014, and 35 acres in 2015. This filamentous cyanobacterium is difficult to eradicate and options for control are being discussed by the TAC.

North Carolina Wildlife Commission (WRC) staff, with help from NC State Parks personnel, collected a large Flathead Catfish (*Pylodictis olivaris*) in June 2014 from the WRC boating access area. This is this first official record of this nonnative, invasive species in the lake, although anecdotal accounts have existed for several years. The mostly likely method of introduction is via human transport and release. The illegal stocking of this predatory catfish will negatively affect sunfish populations and possibly other fishes of Lake Waccamaw. Outreach programs about invasive species are underway at Lake Waccamaw State Park and informational signs have been posted at boat ramps and other high-visibility locations around the lake.

The objective of this study is to conduct standardized surveys for priority species of fish and mollusks in Lake Waccamaw. Species density and/or relative abundance data (catch per unit effort) will be compared to future and previous surveys (Lindquist and Yarbrough 1979, Porter 1985, Mottesi 1998, Shute et al. 2000) to determine if changes in the listing status of these species are warranted, or if management activities need modification.

Methods

Fish surveys

We established four long-term monitoring stations (Figure 1): the southwest shore of the lake near the dam (Site F1), the north shore near Dale's Seafood restaurant (Site F2), the southeast shore near the mouth of Big Creek (Site F3), and the northwest shore near a private residence (added in May 2013; Site F4). Collection locations were georeferenced using handheld GPS units.

Fishes were sampled using 3.1 and 4.6 m seines and the number of seine hauls and duration of sampling were recorded to determine catch per unit effort (CPUE). In addition, we conducted underwater visual surveys at Sites F1 and F3 for Broadtail Madtoms and Waccamaw

Darters, which are not be easily collected using seines or dip nets in vegetated lake habitat. These two sites include large patches of the emergent plant (Maidencane, *Panicum hemitomon*) that provides habitat for the darters and madtoms. CPUE is expressed as the number of individual fish collected per minute of seining (indiv/min), or the number of fish observed per person-minute of visual surveys (indiv/p-min). During seining efforts, 25 individuals of each priority fish species (Table 1) were measured (total length) to assess the size distribution and released. Fish surveys will continue on an annual basis due to the presence of Hydrilla, *Lyngbya*, and Flathead Catfish.

Mollusk surveys

Long-term monitoring surveys were initiated in 2009 at two sites based on the location of previous mussel surveys and our own preliminary work. A third site was added in 2013 due to the treatment of Hydrilla in that area. All surveys are conducted biennially in late-July or early-August (except for the new site that was sampled in May 2013, prior to hydrilla treatment). Site M1 is located on the south shore of the lake near the Lake Waccamaw State Park pier, Site M2 is located on the north shore of the lake near Dale's Seafood restaurant, and Site M3 (added in 2013) is located on the northwest shore near a private residence (Figure 2). The four corners of each site were georeferenced with a handheld GPS unit.

Mussels and snails were collected using a systematic sampling design with random starts (Strayer and Smith 2003; Pooler and Smith 2005) at each 30m X 30m site. Quadrats (0.25 m^2) were sampled at regular distances from a random starting point. The sites were divided into a grid and three random starting points were generated from a random numbers table. Each quadrat that follows a random start is part of the systematic (or sampling) unit. Each random start consisted of 36 quadrats for a total of 108 quadrats per site. The interval between quadrats was 5 m. We collected mussels from the substrate surface and burrowed individuals were also collected at a subset of quadrats (50%) by excavating the substrate to 10 cm. Excavating a portion of the quadrats produces a better estimate of abundance and does not greatly increase sampling effort (Strayer and Smith 2003). Mussels that were found on the surface were recorded separately from those that were burrowed. We measured fifty individuals of each mussel species to assess the size distribution. Mussels were returned to the study site after identification and measurement. Data were analyzed using the USGS Mussel Estimation Program Version 1.5.2 (USGS Leetown Science Center, Aquatic Ecology Lab). This program uses the total counts from the excavated quadrates to calibrate the abundance of the surface counts and will compute population density and abundance estimates. The quantitative study design and the number of quadrats sampled provides mollusk density estimates with low coefficients of variation (except for the very rare species) which will allow for statistical comparisons among surveys. Mussel surveys will continue on a biennial basis due to the establishment of Hydrilla and *Lyngbya* and the application of herbicide for Hydrilla treatment.

In 2013 we began collecting micro-snails (*Amnicola* sp. and *Floridobia* sp.) using a petit ponar grab. Twelve samples were taken randomly within each 30 X 30 study area and washed through a 1.5 mm screen. Micro-snails were then preserved in alcohol for later identification and enumeration.

We recorded habitat conditions at each sampling location. These included substrate composition, water depth (m), water clarity (horizontal Secchi distance in m), temperature (ºC), dissolved oxygen (mg/l), conductivity (µs), and pH. Digital photographs of mussels and sampling locations were taken. Voucher specimens were preserved when necessary and deposited in the NC Museum of Natural Sciences.

Results

Fish surveys

Waccamaw Silversides were present at all sites in all 7 years of our monitoring and the annual mean CPUE for all sites ranged from 23.5 indiv/min of seining (2009) to 2.5 indiv/min (2014; Figure 2). In 2015, the CPUE for all sites was 11.1 indiv/min. The seven-year mean CPUE of Waccamaw Silversides was highest at the south shore of the lake (17.3 indiv/min; Site F1) and lowest at the southeast shore (6.1 indiv/min; Site F2). The mean CPUE of Waccamaw Silversides at the new monitoring site (3 years data; Site F4) was 1.1 indiv/min.

Waccamaw Killifish were also present at all sites in all years with an annual mean CPUE ranging from 0.83 indiv/min (2009) to 2.1 indiv/min (2014; Figure 2). The mean CPUE was similar among sites and ranged from 1.9 indiv/min at Site F4 to 1.1 indiv/min at Site F2. Waccamaw Darters were present at Sites F1 and F3 from 2009-2015 (visual surveys) and were collected with the seines at Sites F2 (2011, 2012, 2014, 2015) and F4 (2014, 2015). Using the visual observations from 2009-2015, the annual mean CPUE for Waccamaw Darters ranged from 1.36 indiv/p-min in 2011 to 0.13 indiv/p-min in 2013 (Figure 2). Waccamaw Darters were most abundant at Site F1 with a mean CPUE of 1.1 indiv/p-min and were rare at Site F3 with a mean CPUE of 0.14 indiv/p-min. Due to high rainfall events in 2013 and 2014 the lake water was darkly stained and underwater visibility was low. This greatly reduced our ability to see Waccamaw Darters during those years. No Broadtail Madtoms were observed (using visual or seining methods) at any of our sites. Since 2009, twenty additional species have been collected during these surveys and the Costal Shiner (*Notropis petersoni*) was the most abundant among these.

Mollusk surveys

The mean density of all mussel species for each year of the study (all sites combined) has been stable over time (ranging from 23.4/m^2 to 25.7/m^2) with the exception of 2013 where the densities increased to 32.9/m^2 (Figure 3). The Waccamaw Spike was the most abundant species at Site M1 (south shore) and the densities range from a high of 11.9/m^2 in 2009 and in 2013 to a low of 9.3/m^2 in 2015 (Table 2). These densities are higher than the Porter (1985) survey results of 6.2/m^2, but lower than the WRC 1997 survey results of 14.4/m^2 (Table 2). Compared to previous surveys, the Ridged Lioplax (*Lioplax subcarinata*) and File Campeloma (*Campeloma limum*) densities are much reduced in this area of the lake (Table 2). The density of micro-snails, from the petit ponar samples, was 111.2/m^2 in 2013 and 218.8/m^2 in 2015 for Waccamaw Snail and 50.2/m^2 in 2013 and 82/m^2 in 2015 for Waccamaw Siltsnail. We collected several species that were not reported during previous surveys at this location, including the Lapped Elimia (*Elimia catenaria dislocata*), Sprite Elimia (*Elimia proxima*), Yellow Lampmussel (*Lampsilis cariosa*), and a mussel in the lance elliptio group (*Elliptio* sp.). The Bayou Physa (*Physella hendersoni*), and the Two-ridge Rams-horn (*Helisoma anceps*) were reported in earlier surveys, but were not located during our surveys.

The Waccamaw Spike was also the most abundant species at site M2 (north shore) and the densities range from of a high of 37.9/m^2 in 2009 to a low of 30.6/m^2 in 2011 which are all higher than reported in the Porter (1985) survey of 25.5/m^2 (Table 3). The remaining species that we encountered were less dense than the Porter (1985) surveys (Table 3). We collected three mussel species that were not encountered during previous surveys in this area which include the Rayed Pink Fatmucket (*Lampsilis splendida*), Savannah Lilliput (*Toxolasma pullus*), and a mussel in the lance elliptio group (*Elliptio* sp.). The density of micro-snails was 86.1/m^2 in 2013 and 132.8/m^2 in 2015 for Waccamaw Snail and 82.5/m^2 in 2013 and 111.2/m^2 in 2015 for Waccamaw Siltsnail.

Site M3 (the new site on the northwest shore) had a density of Waccamaw Spike of $40.2/m^2$ in 2013 and $26.9/m^2$ in 2015 (Table 4). In 2013, the density of the Waccamaw Snail was $10.7/m^2$ and Waccamaw Siltsnail was $21.5/m^2$. Micro-snails were not collected in our samples in 2015 which may be due to the dense patches of Muskgrass (*Chara/Nitella* sp.) which reduced the efficiency of the ponar grab.

Discussion

Fish surveys

The abundance of Waccamaw Silversides has varied over the past 7 years, but they are always collected with minimal effort (CPUE = 2.5 to 23.5 indiv/min) and the population appears resilient. Young-of-the-year silversides comprised the majority of the individuals that we collected which indicates continued successful reproduction and recruitment. The high variability in catch rate is expected due to its schooling behavior and preference for open waters of the lake. CPUE is influenced by how close and how large the nearest schools of silversides are to the sampling crew at the initiation of the seining efforts.

Brook Silversides, a species that has naturally colonized the lake (Moser et al. 1998), was present at Sites F1 and F3. This species was collected very close to the shoreline in or near emergent vegetation. Waccamaw Silversides were typically collected in open waters, and habitat use of the adults and juveniles do not appear to overlap with the Brook Silversides so any competition is likely minimal.

Waccamaw Darters and Waccamaw Killifish were also collected with minimal effort suggesting that abundant populations exist within Lake Waccamaw. Visual surveys for Waccamaw Darters through large patches of emergent vegetation (Maidencane, *Panicum hemitomon*) were more effective for observing this species than seining, but these efforts were hindered by low water visibility in 2013 and 2014. We will continue our annual monitoring surveys of the fish to ensure that populations remain viable during the Hydrilla treatment, and with the presence of Flathead Catfish, and *Lyngbya*.

Mollusk surveys

Priority species of mussels persist in Lake Waccamaw and the population densities of Waccamaw Spike are higher than reported in previous surveys by Porter (1985). Although the densities of some mollusks, especially snails, are lower than reported by Porter (1985) and Mottesi (1998). The densities of Ridged Lioplax and File Campeloma at our study sites are much lower than previous surveys and the reasons for the decline are not known. However, our sampling technique for snails is not as efficient as a suction dredge that was used in the Porter surveys.

The density and richness of mollusks is higher at the northern end of the lake (Site M2 and M3) and this may be linked to the finer substrate composition with more organic material that is located in that area. The Savannah Lilliput was found at Site M2, which is a significant collection due to its rarity in North Carolina. This species was collected in very low numbers in the Porter surveys, and had not been collected again in Lake Waccamaw until our surveys in 2009. We will continue our biennial monitoring surveys of the mollusks to ensure that populations remain viable during the Hydrilla treatment and with the presence of *Lyngbya*.

Acknowledgments

We thank numerous individuals from the North Carolina Wildlife Resources Commission, North Carolina State Parks, North Carolina Department of Transportation, North Carolina Museum of Natural Science, and volunteers for providing invaluable assistance while sampling in Lake Waccamaw. Ed Corey and Tyler Black provided helpful comments and suggestions to previous drafts of the report.

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Figure 1: Map of Lake Waccamaw indicating Fish (F1 – F4) and mollusk (M1 – M3) monitoring sites.

Figure 2. Mean catch per unit effort (CPUE) of Waccamaw Silversides, Killifish (#/min of seining) and Waccamaw Darters (#/person-min of visual observation) among years. Vertical bars represent standard deviation.

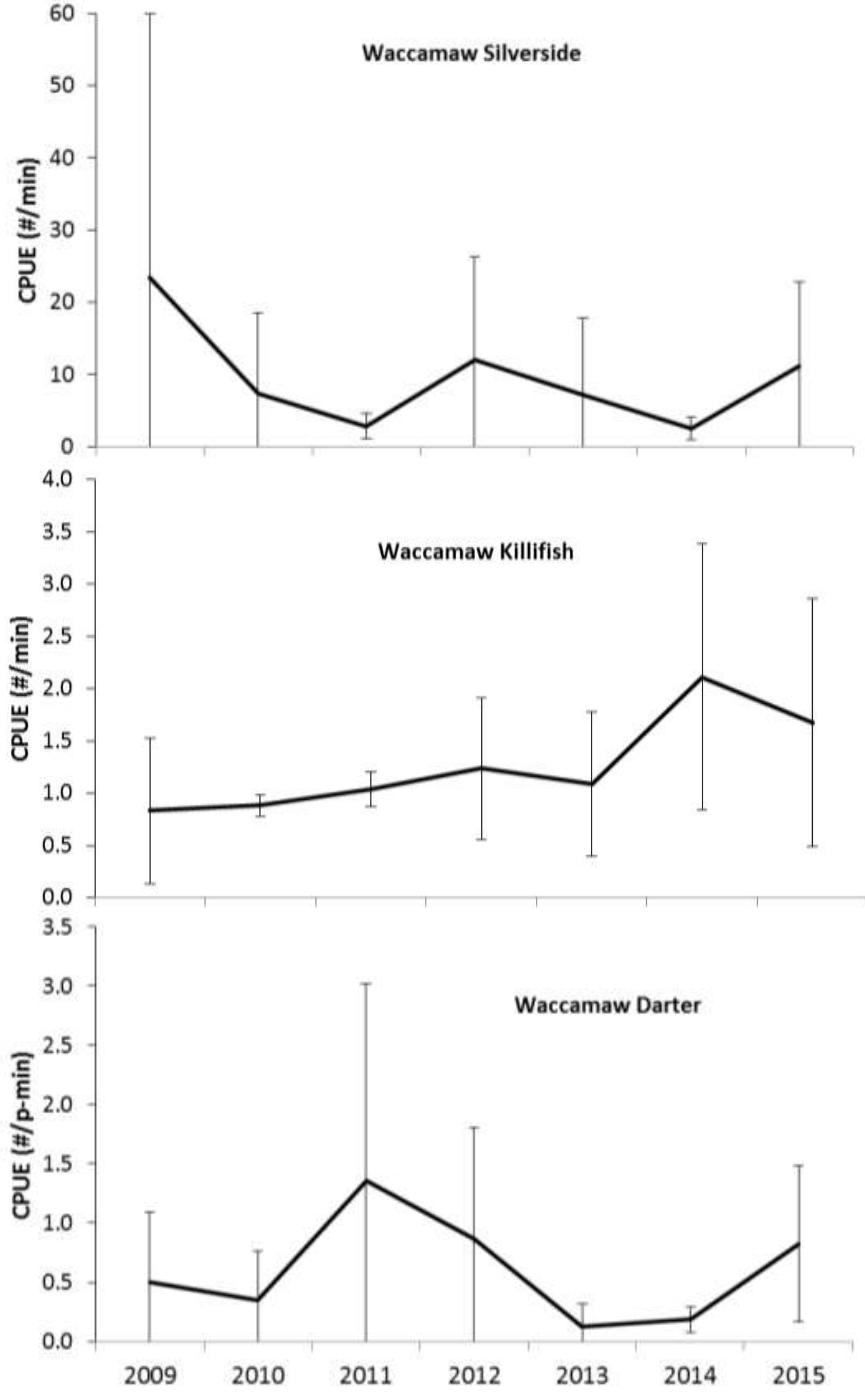

Figure 3. Annual mean density (#/m^2) of all species of mussels at monitoring sites from 2009-2015 in Lake Waccamaw.

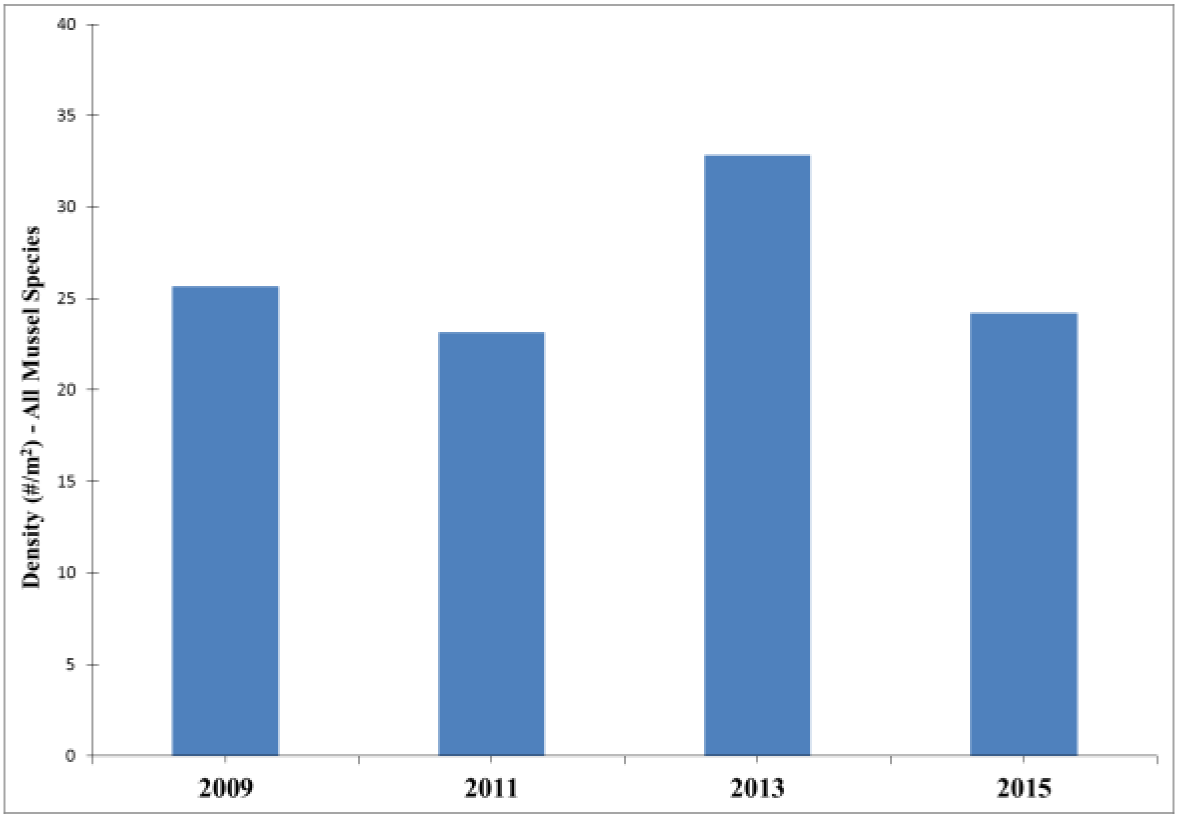

Table 1. Rare species located in Lake Waccamaw and their federal and state listing status.

Scientific Name	Common Name	Taxa	Federal Status	NC Status
Etheostoma perlongum	Waccamaw Darter	fish	FSC	T
Fundulus waccamensis	Waccamaw Killifish	fish	FSC	SC
Menidia extensa	Waccamaw Silverside	fish	T	T
Noturus sp. 2	Broadtail Madtom	fish	FSC	SC
Elliptio folliculata	Pod Lance	mussel		SC
Elliptio waccamawensis	Waccamaw Spike	mussel	FSC	E
Lampsilis cariosa	Yellow Lampmussel	mussel	FSC	E
Lampsilis fullerkati	Waccamaw Fatmucket	mussel	FSC	T
Lampsilis radiata	Eastern Lampmussel	mussel		T
Lampsilis splendida	Rayed Pink Fatmucket	mussel		SR
Leptodea ochrachea	Tidewater Mucket	mussel		T
Toxolasma pullus	Savannah Lilliput	mussel	FSC	E
Villosa delumbis	Eastern Creekshell	mussel		SR
Amnicola sp.	Waccamaw Snail	snail		SC
Floridobia sp.	Waccamaw Siltsnail	snail		SC
Ferrissia hendersoni	Blackwater Ancylid	snail		SC
Lioplax subcarinata	Ridged Lioplax	snail		Watch list
Viviparus intertextus	Rotund Mysterysnail	snail		SR

Table 2. Mollusk density and coefficient of variation at Site M1 located on the south shore of Lake Waccamaw at the Lake Waccamaw State Park.

Species		Density #/m^2 (CV %)					
		Porter 1985	WRC 1997	WRC 2009	WRC 2011	WRC 2013	WRC 2015
All mussels combined				12.44	11.398	12.785	10.228
Elliptio waccamawensis	Waccamaw Spike	6.2 (107.3)	14.4	11.9 (6.6)	10.7 (6.5)	11.9 (7.3)	9.3 (8.0)
Leptodea ochrachea	Tidewater Mucket	1.9 (140.0)	2.8	0.30 (41.7)	0.58 (27)	0.30 (39.0)	0.63 (35.3)
Lampsilis fullerkati	Waccamaw Fatmucket	0.15 (466.7)		0.22 (49.3)	0.11 (79)	0.08 (138)	0.07 (69.6)
Lampsilis cariosa	Yellow Lampmussel					0.49 (32.7)	0.19 (61.5)
Elliptio spp. (lance)	lance elliptio group			0.04 (139.3)			
Snails							
Lioplax subcarinata	Ridged Lioplax	8.5 (128.0)	21.1	0.59 (34.0)	0.04 (98.5)		
Campeloma limum	File Campeloma	29.3 (127.3)	9.82	0.44 (42.9)	2.2 (89.1)		
Elimia catenaria dislocata	Lapped Elimia			0.04 (135.4)			
Elimia proxima	Sprite Elimia				0.04 (98.5)		
Helisoma anceps	Two-ridge Rams-horn		0.12	0.04 (98.5)			

Table 3. Mollusk density and coefficient of variation at Site M2 located on the north shore of Lake Waccamaw, near Dale's Seafood restaurant.

Species		Density #/m^2 (CV %)				
		Porter 1985	WRC 2009	WRC 2011	WRC 2013	WRC 2015
All mussels combined			41.5	33.9	38.9	34.9
Elliptio waccamawensis	Waccamaw Spike	25.5 (110.2)	37.9 (7.3)	30.6 (4.4)	36.8 (4.4)	32.8 (5.4)
Leptodea ochrachea	Tidewater Mucket	5.3 (118.0)	2.5 (16.3)	2.21 (15.3)	1.44 (16.4)	0.54 (37.5)
Lampsilis fullerkati	Waccamaw Fatmucket	1.5 (308.1)	0.67 (26.7)	0.63 (27)	0.11 (65.4)	0.33 (31.9)
Lampsilis splendida	Rayed Pink Fatmucket		0.19 (61.8)	0.15 (65.2)		0.08 (100)
Lampsilis cariosa	Yellow Lampmussel	0.16 (512.5)	0.11 (46.4)	0.22 (49.1)	0.44 (23.1)	1.1 (28.5)
Toxolasma pullus	Savannah Lilliput		0.07 (69.6)			
Elliptio spp. (lance)	lance elliptio group		0.07 (98.5)	0.08 (98.5)	0.08 (69.3)	0.04 (98.5)
Snails						
Campeloma limum	File Campeloma	21.2 (157.3)	2.3 (19.5)	0.45 (38.1)		0.08 (98.8)
Lioplax subcarinata	Ridged Lioplax	11.3 (118.5)	2.0 (24.8)	0.4 (47.4)	0.04 (138)	
Helisoma anceps	Two-ridge Eams-horn		0.04 (98.5)	0.04 (138)		

Table 4. Mollusk density and coefficient of variation at Site M3 located on the northwest shore of Lake Waccamaw.

Species		Density #/m^2 (CV %)	
		WRC 2013	WRC 2015
All mussels combined		44.3	28.5
Elliptio waccamawensis	Waccamaw Spike	40.2 (4.3)	26.9 (5.7)
Leptodea ochrachea	Tidewater Mucket	0.41 (31.5)	0.37 (38.1)
Lampsilis fullerkati	Waccamaw Fatmucket	1.0 (23.5)	0.51 (38.1)
Lampsilis splendida	Rayed Pink Fatmucket	0.19 (71.6)	
Lampsilis cariosa	Yellow Lampmussel	2.1 (12.9)	0.63 (29.5)
Elliptio spp. (lance)	lance elliptio group	0.37 (41.0)	0.04 (139.3)

State of Vermont
Department of Environmental Conservation
Aquatic Nuisance Control Permit Program
10 V.S.A. Chapter 47 § 1263a

Re: Lake St. Catherine Association, Applicant/Permittee
c/o Jeff Crandall
227 Fox Meadow Road
Scarsdale, NY 10583

Application No. 2001-C08
District ID. #RU01-0255

Aquatic Control Technology, Inc., Co-Applicant/Co-Permittee
c/o Gerald Smith
11 John Road
Sutton, MA 01590

Project: Use the aquatic herbicide, Sonar A.S. to control Eurasian watermilfoil in Lake St. Catherine, Lily Pond and Little Lake in Poultney and/or Wells, Vermont

BACKGROUND

On December 10, 2001, the Vermont Department of Environmental Conservation (Department) received a complete application from the Lake St. Catherine Association (Applicant and Permittee) and Aquatic Control Technology, Inc. (Co-Applicant and Co-Permittee) seeking a permit to use an aquatic pesticide under the provisions of 10 V.S.A. § 1263a, Aquatic Nuisance Control Permits, to control Eurasian watermilfoil (watermilfoil) in Lake St. Catherine, Lily Pond and Little Lake located in Poultney and/or Wells, Vermont. The Applicant and Co-Applicant propose a five-year integrated management program, employing both chemical and non-chemical control techniques. Use of the aquatic herbicide Sonar* A.S. (* Trademark of SePRO Corporation, Carmel, IN), active ingredient fluridone, is proposed in year one of the program. The Applicant and Co-Applicant originally proposed a target initial fluridone concentration of 10 parts per billion (ppb). Due to the Department's concern regarding potential negative impacts on the non-target environment, the Applicant and Co-Applicant revised the request to propose a target initial fluridone concentration of 8 ppb.

On December 14, 2001, the Department notified state and local officials and others having an interest in the project of the application and provided an opportunity to file written comments or to request a public information meeting. The Department provided written notice in the legal classified section of the Rutland Herald on December 14, 2001 with comments or requests for a public information meeting due on December 31, 2001. No requests for a public information meeting were received.

The Findings that support the decision and conditions of this permit can be found on pages 11 through 26.

DECISION AND PERMIT

Based upon the Findings presented on pages 11 through 26 and supporting documents on file with the Department, it is the decision of the Department that the requested use of the aquatic herbicide

Sonar A.S. in year one of a five-year integrated management plan to control Eurasian watermilfoil in Lake St. Catherine, Lily Pond and Little Lake in Poultney and/or Wells, Vermont is in conformance with 10 V.S.A. § 1263a.

In accordance with 10 V.S.A. § 1263a(e), the Lake St. Catherine Association (Applicant and Permittee) and Aquatic Control Technology, Inc. (Co-Applicant and Co-Permittee) are authorized to use Sonar A.S. in Lake St. Catherine, Lily Pond and Little Lake in compliance with the following conditions. Unless otherwise specified, the term "treatment" in these conditions refers to the initial treatment of the waterbodies with Sonar A.S. and any or all booster treatments.

1. This permit is valid upon signing and shall expire five years from the date of signing.

2. In one of the years 2004, 2005 or 2006 the Permittee and Co-Permittee are authorized to:

    a. After the establishment of a thermocline in Lake St. Catherine, conduct a whole-lake treatment of Lake St. Catherine, Lily Pond and Little Lake between May 1 and June 15 with the aquatic herbicide, Sonar A.S., active ingredient fluridone, EPA Registration No. 67690-4, to achieve a target in-lake fluridone concentration of 8 parts per billion in each waterbody; and

    b. Conduct follow-up booster treatments with Sonar A.S. at a target fluridone concentration of up to 8 parts per billion, only if needed to establish or re-establish the target concentration of 8 parts per billion and/or maintain fluridone levels of 5 to 8 parts per billion in each waterbody.

3. The Permittee and Co-Permittee shall maintain fluridone levels in each waterbody of at least 5 parts per billion for a minimum of 90 days following the initial treatment.

4. No booster treatments with Sonar A.S. shall occur after October 31 of the treatment year.

5. In the treatment year, the sum of all treatments shall not exceed 150 parts per billion.

6. The specific product used, Sonar A.S., must be registered with the Vermont Agency of Agriculture, Food and Markets for use in Vermont at the time of the treatment, and shall be applied in full conformance with all label requirements and state and federal regulations in effect at the time of the treatment.

7. The Permittee and Co-Permittee shall obtain water column temperature profile measurements in Lake St. Catherine within the 48-hour period prior to each treatment to aid in calculating water volume and determining the amount of Sonar A.S. required. The Permittee and Co-Permittee shall use data from the bathymetric study, dated 9/28/01 and submitted with the application, to calculate water volume.

8. The disposal of surplus Sonar A.S., container rinseate, and empty product containers shall be conducted according to product label requirements and federal and state law and regulations.

9. Sonar A.S. shall only be applied by a pesticide applicator certified by the Vermont

Agency of Agriculture, Food and Markets in Category Five - Aquatics, and only by a Co-Permittee of this permit. Sonar A.S. shall only be applied in the presence of someone with prior experience in its application.

10. The Permittee and Co-Permittee shall submit to the Department an herbicide application record form (Attachment A) along with water column temperature profile measurements and chemical treatment quantity calculations associated with each treatment within seven calendar days following the date of each treatment of Lake St. Catherine, Lily Pond and/or Little Lake with Sonar A.S.

11. Prior to any Sonar A.S. treatment taking place, the Permittee and Co-Permittee shall submit to the Department the name(s), current address, and telephone number of all owners of property bordering Mill Brook and/or Wells Brook between Geer Road and Route 149.

12. Aquatic Nuisance Control Permit #2000-H03 authorizing mechanical harvesting in Lake St. Catherine, Lily Pond and Little Lake shall be revoked on the date that Public Notification as described in Permit Condition No. 28a of this permit is initiated. On that date, the Permittee shall submit to the Department acknowledgement of revocation of Permit #2000-H03.

13. A duly authorized representative(s) of the Department may at any time inspect the project, including the operation and maintenance thereof.

14. The Permittee shall demonstrate to the Department how each year of the approved five-year integrated management plan will be funded. Written approval of the funding strategy must be obtained from the Department prior to any Sonar A.S. treatment taking place.

15. The Permittee shall meet with the Department on an annual basis to discuss the level of watermilfoil control achieved/maintained, the impacts to non-target species, and other pertinent issues as well as the most effective strategy to be implemented as the next phase of the five-year integrated management plan. The Permittee shall implement each phase of the integrated management plan as mutually agreed upon by the Department and the Permittee at the annual meeting and shall not change the management plan without prior written approval from the Department. The Permittee's obligations under this condition shall continue until the five-year integrated management plan is completed, regardless of the expiration date of this permit.

16. The Permittee shall obtain Aquatic Nuisance Control Permits for those chemical and non-chemical components of the five-year integrated management plan that require permits pursuant to 10 V.S.A. 1263a (e.g., spot or partial-lake aquatic herbicide treatment, bottom barrier), prior to implementing those components.

17. The Permittee shall maintain all data and records relating to the activities authorized by this permit and the associated five-year integrated management plan for a period of one year following the completion of the integrated management plan. The Permittee's obligations under this condition shall continue until the one-year period has passed, regardless of the expiration date of this permit. The Co-Permittee shall maintain all data and records relating to the Co-Permittee's obligations under this permit for a period of two years following completion of the Sonar A.S. treatment.

18. There shall be **no use** of Lake St. Catherine, Lily Pond, Little Lake and Mill Brook north of Geer Road for any purpose on the day(s) of a Sonar A.S. treatment of any of the lakes, which includes but is not limited to:

    - swimming/wading
    - boating
    - fishing
    - irrigation
    - direct water intakes used for drinking or any other purpose, including toilet flushing.

    If an individual chooses to ignore this restriction, he or she does so at his or her own risk.

19. There shall be **no irrigation use** of Lake St. Catherine, Lily Pond and Little Lake water and water from their outlet stream (Mill Brook) north of Geer Road, including use for watering lawns, trees, shrubs or plants, beginning the day of a Sonar A.S. treatment of Lake St. Catherine, Lily Pond or Little Lake and continuing until the Department provides notification to the Permittee that the restriction has been lifted. If an individual chooses to ignore this irrigation use restriction he or she does so at his or her own risk. [The Department intends to base lifting the irrigation use restriction on chemical analyses of representative water samples, as specified in conditions 25 through 27 below, that indicate that the concentration of fluridone is less than 5 parts per billion.]

20. There shall be **no domestic use** of Lake St. Catherine, Lily Pond and Little Lake water and water from Mill Brook north of Geer Road, beginning the day of a Sonar A.S. treatment of Lake St. Catherine, Lily Pond or Little Lake and continuing until the Department provides notification to the Permittee that the restriction has been lifted. The only exception to this restriction shall be that direct water intakes may be used for flushing toilets at the beginning of the first day following the completion of a treatment. If an individual chooses to ignore this domestic use restriction he or she does so at his or her own risk. [The Department intends to base lifting the domestic use restriction on chemical analyses of representative water samples, as specified in conditions 25 through 27 below, that indicate that (1) the concentration of fluridone is less than or equal to 20 parts per billion and (2) n-methylformamide (NMF) is below the detection limit of 2 parts per billion.]

21. **Swimming/wading** in Lake St. Catherine, Lily Pond, Little Lake and Mill Brook north of Geer Road may resume 24 hours after the completion of a Sonar A.S. treatment of Lake St. Catherine, Lily Pond or Little Lake.

22. **Boating** on Lake St. Catherine, Lily Pond, Little Lake and Mill Brook north of Geer Road may resume at the beginning of the first day following the completion of a Sonar A.S. treatment of Lake St. Catherine, Lily Pond or Little Lake.

23. **Fishing** in Lake St. Catherine, Lily Pond, Little Lake and Mill Brook north of Geer Road may resume at the beginning of the first day following the completion of a Sonar A.S. treatment of Lake St. Catherine, Lily Pond or Little Lake.

24. The Permittee shall supply bottled water for the duration of the required domestic use restriction periods to all persons restricted from using their domestic water supply due to the above use restrictions, unless other arrangements are made by those affected.

25. The Permittee and Co-Permittee shall collect water from at least nine sites in Lake St. Catherine, one site in Lily Pond and two sites in Little Lake, and from at least one site in Mill Brook approximately one mile from the outlet of Little Lake. Samples for fluridone analysis shall be collected beginning approximately 24 hours after completion of each Sonar A.S. treatment and continuing weekly until at least 90 days after the initial treatment. Weekly sampling at all sites shall continue after 90 days if the fluridone concentration is 5 parts per billion or above at any site, until the fluridone concentration falls below 5 parts per billion at all sites. Sampling at one or more sites may be discontinued prior to this time if the Permittee and Co-Permittee receive prior written approval from the Department to discontinue the sampling.

26. The Permittee and Co-Permittee shall also collect water from one site in Mill Brook approximately one-quarter mile from the outlet of Little Lake. Samples for fluridone analysis shall be collected from this site beginning approximately 24 hours after completion of each Sonar A.S. treatment and continuing weekly until all use restrictions except the irrigation use restriction have been lifted by the Department.

27. Water samples collected in accordance with conditions 25 and 26 above shall be analyzed at the SePRO Corporation laboratory for fluridone concentration by the FasTEST method or by another method or laboratory approved by the Department. Water samples shall also be collected and properly preserved for potential n-methylformamide (NMF) analysis at the same times and same sites as the fluridone samples are collected. If the sample results indicate a fluridone concentration at or above 30 parts per billion, the Permittee and Co-Permittee shall analyze at least one preserved water sample from the site with the highest fluridone concentration for NMF. All sampling results shall be submitted to the Department as soon as they become available to the Permittee and Co-Permittee. Prior to treatment, the Permittee and Co-Permittee shall submit to the Department (1) a map of Lake St. Catherine, Lily Pond, Little Lake and Mill Brook with the sampling locations identified, and (2) the name of the laboratory that will conduct the analyses for NMF if such analyses are required. Individuals collecting water samples for fluridone using the FasTEST method shall be trained by SePRO Corporation. Written approval of the sampling locations must be obtained from the Department prior to any Sonar A.S. treatment taking place. Additional sampling locations and samples may be required in Mill Brook if sample results from the approved Mill Brook sampling sites show fluridone levels above 5 parts per billion.

28. The Permittee shall conduct public notification in the following manner:

Initial Sonar A.S. Treatment of Lake St. Catherine, Lily Pond and Little Lake

a. An informational flier shall be hand-delivered to all owners of property abutting Lake St. Catherine, Lily Pond, Little Lake or Mill Brook north of Geer Road, or shall be mailed to property owners if the property is vacant, at least 30 days prior to the initial Sonar A.S. treatment. The flier shall state:

g The proposed dates of the initial and potential booster treatments;
g The aquatic herbicide to be used;

- g That all of Lake St. Catherine, Lily Pond and Little Lake will be treated and water use restrictions will affect all three waterbodies and Mill Brook north of Geer Road;
- g The water use restrictions and the time period the restrictions will be in effect, including conditions upon which domestic use and irrigation use can resume;
- g That signs posted along roadways will provide specific use restriction dates;
- g That bottled water will be provided, if requested, to any person restricted from using their domestic water supply;
- g That the irrigation use restriction on the label for the herbicide specifically warns: "For tobacco, tomatoes, peppers or other plants within the Solanaceae family and newly seeded crops or newly seeded grasses such as over-seeded golf course greens, do not use Sonar A.S. treated water if measured fluridone concentrations are greater than 5 ppb. Furthermore, when rotating crops, do not plant members of the Solanaceae family in land that has been previously irrigated with fluridone concentrations in excess of 5 ppb;" and
- g The contact name(s), address(es), and telephone number(s) for the Permittee for further information.

The flier shall also state that notification of the exact treatment dates will be posted in the locations described in condition 28c below and submitted to local radio stations for public service announcements. The flier shall inform property owners in bold print that if a residence or cottage will be rented at any time after the treatment and prior to December 31 of the year in which the treatment occurred, the property owner is responsible for informing all tenants of the treatment and the water use restrictions. A copy of the flier shall be provided to the Department.

b. The same informational flier described in condition 28a above shall be provided at least 30 days prior to the initial Sonar A.S. treatment to any commercial camps that use the treated waters or Mill Brook north of Geer Road, and shall be provided, prior to the children attending, to all parents of children who will be attending the camps in the season of treatment.

c. Signs shall be posted (1) facing the road, at least once every mile along the lake side of the roadways in the vicinity of Lake St. Catherine, Lily Pond, Little Lake and Mill Brook north of Geer Road; (2) at all public and private campgrounds and access points where the public might enter or use Lake St. Catherine, Lily Pond, Little Lake or Mill Brook north of Geer Road; and (3) at the Poultney and Wells town offices. The signs at locations (1) and (2) shall be at least 24 inches in height by 18 inches in width. The signs at location (3) shall be 11 inches in height by 8.5 inches in width. All signs shall be made of brightly colored waterproof paper and printed with waterproof ink. The signs shall state:

NOTICE
AQUATIC HERBICIDE IN USE

Lake St. Catherine, Lily Pond and Little Lake were treated with the aquatic herbicide Sonar A.S. on ________________
date

NO Drinking or Household Use UNTIL FURTHER NOTICE
(EXCEPT flushing toilets may resume on ________)
date

NO Use for Irrigation or Plant Watering UNTIL FURTHER NOTICE

NO Swimming/Wading for 24 hours after completion of the treatment
(may resume at ________on ________)
time date

NO Boating or Fishing Until _________
date

Any individual who chooses to ignore these restrictions does so at their own risk.

For information contact: ____________________________________
Permittee contact name/telephone number

The signs shall be posted at least 24 hours prior to the initial Sonar A.S. treatment. A representative copy of the sign shall be provided to the Department. When the domestic water use restriction has been lifted, the signs shall be updated to indicate that drinking and household use may resume as of the date specified by the Department. The signs shall remain posted until replaced by new signs. The Permittee shall remove the signs upon their replacement.

Booster Sonar A.S. Treatments of Lake St. Catherine, Lily Pond and Little Lake

d. The same signs described in condition 28c above, but specifying the date of the booster treatment and the new restriction dates, shall be posted at least 24 hours prior to each Sonar A.S. booster treatment of Lake St. Catherine, Lily Pond or Little Lake, replacing the signs announcing the initial treatment or a previous booster treatment. The signs shall be the same type and size(s) and posted at the same locations as the signs described in condition 28c above. A representative copy of the sign shall be provided to the Department. When the domestic water use restriction has been lifted, the signs shall be updated to indicate that drinking and household use may resume as of the date specified by the Department. The signs shall remain posted until replaced by new signs. The Permittee shall remove the signs upon their replacement.

All Sonar A.S. Treatments

e. Public service announcements shall be submitted to local radio stations to be aired daily beginning 48 hours prior to each Sonar A.S. treatment and ending no earlier than two days following the day all use restrictions have been lifted. The announcements shall contain the same information as stated in condition 28a above but shall specify the exact date(s) of the most recent treatment of the lakes. The announcements shall change (1) for each treatment and (2) when all restrictions have been lifted and all normal uses of Lake St. Catherine, Lily Pond, Little Lake and Mill Brook north of Geer Road are again allowed.

Following the Sonar A.S. Treatments

f. When all use restrictions have been lifted by the Department from Lake St. Catherine, Lily Pond, Little Lake, and Mill Brook, the Permittee shall replace the signs described in conditions 28c and/or 28d above with signs of the same type and size(s) and posted at the same locations that state that the aquatic herbicide treatment of Lake St. Catherine, Lily Pond and Little Lake is completed, all water use restrictions have been lifted, and all normal uses of Lake St. Catherine, Lily Pond, Little Lake and Mill Brook are again allowed. These signs shall remain posted for a minimum of two weeks. The Permittee shall remove the signs after the two-week period has passed. The signs shall remain posted for no longer than four weeks.

29. If sample results indicate fluridone levels above 5 parts per billion may be occurring in Mill Brook south of Geer Road, upon notification from the Department the Permittee shall immediately contact by telephone or in person all owners of property bordering Mill Brook or Wells Brook between Geer Road and Route 149 and inform them that there shall be **no irrigation use** of Mill Brook and Wells Brook between Geer Road and Route 149, including use for watering lawns, trees, shrubs or plants, until the Permittee provides notification to the property owner that the restriction has been lifted. If an individual chooses to ignore this irrigation use restriction he or she does so at his or her own risk. [The Department will notify the Permittee of when the irrigation use restriction may be lifted and property owners may be notified.]

30. The Permittee shall submit an annual report on Lake St. Catherine, Lily Pond and Little Lake to the Department in the year of Sonar A.S. treatment and for four consecutive years thereafter on or before December 31 of each year. An extension of time may be granted for cause. A request for an extension must be received by the Department prior to the December 31 due date. The annual report shall include (a) a qualitative assessment of the status of Eurasian watermilfoil growth and distribution in each waterbody and in the open water areas of the Class II wetlands; (b) a map of each waterbody and the Class II wetlands with Eurasian watermilfoil growth and distribution depicted; (c) a detailed description of all follow-up control methods used in the waterbody and an assessment of the success of each method; and (d) a discussion of the educational efforts implemented as part of the five-year integrated management plan. The Permittee's obligations under this condition shall continue until the five-year integrated management plan is completed, regardless of the expiration date of this permit.

31. The Permittee shall conduct a pre-treatment qualitative aquatic plant survey of Lake St. Catherine, Lily Pond and Little Lake in August no more than three years prior to the initial treatment. The Permittee shall submit a report on the pre-treatment plant survey to the Department prior to any Sonar A.S. treatment occurring. The report shall include at a minimum (a) the date(s) of the pre-treatment survey; (b) the names of survey crew members; (c) a list of aquatic plant species present and their abundance, with specific information on *Brasenia schreberi*, *Ceratophyllum demersum*, *Elodea canadensis*, *Eleocharis robbinsii*, *Isoetes tuckermanii*, *Megalodonta beckii*, *Myriophyllum farwellii*, *Myriophyllum verticillatum*, *Nuphar variegata*, *Nymphaea odorata*, *Najas flexilis*, *Najas gracillima*, *Potamogeton crispus*, *Potamogeton strictifolius*, and *Utricularia gibba*, if identified on the

survey; and (d) a map depicting specific areas surveyed, with associated text describing species present and their abundance (include abundance key) for each area.

32. The Permittee shall conduct two post-treatment qualitative aquatic plant surveys of Lake St. Catherine, Lily Pond, Little Lake, and the open water areas in the Class II wetlands located 1) at the north and south ends of Lily Pond as scrub shrub and emergent wetlands, 2) along the inlet to Little Lake from Lake St. Catherine as scrub shrub and emergent wetlands, and 3) at the south end of Little Lake as scrub shrub and forested wetlands, using the same survey methods and during approximately the same time period each year as the pre-treatment survey described above in condition 31 for comparative purposes. The first post-treatment plant survey shall occur in the year of treatment and the second in the year following treatment of Lake St. Catherine, Lily Pond and Little Lake. The Permittee shall submit a report to the Department within 45 calendar days following each post-treatment plant survey. The report shall include at a minimum (a) the date(s) of the post-treatment survey; (b) the names of survey crew members; (c) treatment effectiveness on Eurasian watermilfoil; (d) general impacts on non-target aquatic plants in each waterbody and associated Class II wetland areas and specific information on impacts on *Brasenia schreberi*, *Ceratophyllum demersum*, *Elodea canadensis*, *Eleocharis robbinsii*, *Isoetes tuckermanii*, *Megalodonta beckii*, *Myriophyllum farwellii*, *Myriophyllum verticillatum*, *Nuphar variegata*, *Nymphaea odorata*, *Najas flexilis*, *Najas gracillima*, *Potamogeton crispus*, *Potamogeton strictifolius*, and *Utricularia gibba*; and (e) a map depicting specific areas surveyed, with associated text describing species present and their abundance (include abundance key) for each area.

33. The Permittee and Co-Permittee shall conduct the Sonar A.S. treatment and the associated five-year integrated management plan in strict accordance with the permit application dated November 12, 2001 and received November 13, 2001; Co-Applicant signature dated November 9, 2001 and application materials received from ACT, Inc. on December 3, 2001; information received from Bob Moore on December 10, 2001; the following Findings; and the conditions of this permit, with such minor modifications as may be approved in writing by the Department.

34. In the event that Aquatic Control Technology, Inc. represented herein by Gerald Smith is not the project applicator, the new project applicator shall become the Co-Permittee, submit the required documentation (see Attachment B) to the Department, and receive written authorization from the Department to become the Co-Permittee before performing any and all activities required of the Co-Permittee under this permit.

35. This permit may be modified for cause upon written request for modification that contains facts or reasons supporting the request, or upon the Department's own motion. If the Department determines that modification is appropriate, only the conditions subject to modification shall be reopened. Any modification under this condition shall be performed in accordance with the public notice requirements of the *Public Review and Comment Procedures for Aquatic Nuisance Control Permit Applications and General Permits* under 10 V.S.A. § 1263a dated January 30, 2003 and approved by the Secretary of the Agency of Natural Resources on February 18, 2003. Cause for modification of this permit includes, but shall not be limited to:

    a. Alterations to the activities authorized by this permit which occurred after permit issuance and which justify the application of conditions that are different or absent in the existing permit; or

b. The receipt of information concerning the activities authorized by this permit which was not available at the time the permit was issued and which would have justified different permit requirements at the time of permit issuance.

36. After notice and opportunity for a hearing, this permit may be suspended or revoked for cause in whole or in part, upon a written request for suspension or revocation which contains facts or reasons supporting the request, or upon the Department's own motion. Cause for suspension or revocation includes:

    a. Violation of any of the terms or conditions of this permit;

    b. Failure by the Permittee or Co-Permittee to disclose all relevant facts during the permit issuance process that were known at that time;

    c. Misrepresentation of any relevant fact at any time during the permit issuance process;

    d. A determination by the Department that a reasonable non-chemical alternative is available;

    e. A determination by the Department that the risk to the non-target environment resulting from the activities authorized by this permit is unacceptable;

    f. A determination by the Department that the risk to public health resulting from the activities authorized by this permit is more than negligible; or

    g. A determination by the Department that this activity does not provide a public benefit.

37. Nothing in this permit shall be construed to relieve the Permittee, Co-Permittee or their agent(s) from civil or criminal penalties for noncompliance with the conditions of this permit.

38. Nothing in this permit shall be construed to preclude the institution of any legal action or relieve the Permittee, Co-Permittee or their agent(s) from any responsibility, liabilities or penalties established pursuant to any applicable federal, state and local laws, regulations, or permits.

39. Issuance of this permit does not convey any property rights in either real or personal property, or any exclusive privileges, nor does it authorize any injury to private property or any invasion of personal rights, nor any infringement of federal, state or local laws or regulations.

40. The provisions of this permit are severable, and if any provision of this permit, or the application of any provision of this permit to any circumstance, is held invalid, the application of such provision to other circumstances, and the remainder of this permit, shall not be affected thereby.

41. If a permit renewal is desired, an application should be filed at least 90 days prior to the expiration date of this permit. A decision to issue or deny a permit will be based on the relevant statutory criteria and Department rules, procedures and policies prevailing at that time.

FINDINGS

The Department has reviewed all the information received from the Lake St. Catherine Association (Applicant) and Aquatic Control Technology, Inc. (Co-Applicant) relative to the application and makes the following Findings as required under 10 V.S.A. § 1263a(e).

1. Jurisdiction

Lily Pond is located in Poultney, Vermont. Little Lake is located in Wells, Vermont. Lake St. Catherine is located in both Poultney and Wells, Vermont. All three waterbodies are designated as waters of the state. Since the proposed activity is to use an aquatic pesticide to control an aquatic nuisance in these waters, the Secretary of the Agency of Natural Resources has jurisdiction under 10 V.S.A. § 1263a. Further, 10 V.S.A. § 1263a(e) directs the Secretary to issue a permit for pesticide use when the Secretary can make the following five findings:

1) There is no reasonable non-chemical alternative available;
2) There is acceptable risk to the non-target environment;
3) There is negligible risk to public health;
4) A long-range management plan has been developed which incorporates a schedule of pesticide minimization; and
5) There is a public benefit to be achieved from the application of the pesticide, or in the case of a pond located entirely on a landowner's property, no undue adverse effect upon the public good.

The Secretary has designated the Commissioner of the Department of Environmental Conservation or the Commissioner's designated representative to act on the Secretary's behalf in the issuance or denial of these permits.

2. General Description

Collectively, Lake St. Catherine with Lily Pond to the north and Little Lake to the south are approximately 1,088 acres in size. Lily Lake is 22 acres with an average depth of 4 feet; Lake St. Catherine is 904 acres with an average depth of 37 feet; and Little Lake is 162 acres with an average depth of 4 feet. This system of three lakes drains into Mill Brook and ultimately into Lake Champlain.

The Department first confirmed watermilfoil (*Myriophyllum spicatum*) in Lake St. Catherine, Lily Pond and Little Lake in 1983. Watermilfoil is an aggressive, nonnative, aquatic plant that has the ability to completely invade lakes once introduced. The Department conducted a watermilfoil survey of Lake St. Catherine, Lily Pond and Little Lake in July 2000. At the time of the 2000 survey, watermilfoil was found at varying densities from uncommon to very abundant around all of the shores of these waterbodies, extending to varying depths within the 34 surveyed areas. A 2001 report prepared by the Co-Applicant and ReMetrix estimated watermilfoil coverage in 30% of the total surface area of the three lakes combined. There were 120 acres of abundant cover extending along 90% of the shoreline of Lake St. Catherine and another 200 acres of common and scattered cover in the northern and southern ends of Lake St. Catherine as well as Lily Pond and Little Lake. In addition to watermilfoil, 27 aquatic plant species were documented. The report made note that the only aquatic plant species of special concern seen was humped bladderwort (*Utricularia gibba*) in Little Lake. The report also recorded a submersed spikerush (*Eleocharis*) that may have been Robbins spikerush (*Eleocharis robbinsii*) at two data point locations in Little Lake, but insufficient plant samples were collected for positive taxonomic identification.

Aquatic plant surveys conducted by the Department documented the presence of three rare plants in Lake St. Catherine, Farwell's watermilfoil (*Myriophyllum farwellii*) in 1984; slender naiad (*Najas gracillima*) in 2003; and straight-leaved pondweed (*Potamogeton strictifolius*) in 1989 and 1999. The Department also documented one uncommon species, humped bladderwort (*Utricularia gibba*), in Lily Pond in 1998 and in Lake St. Catherine in 1990 and 2003. The Nongame and Natural Heritage Program of the Vermont Department of Fish and Wildlife has records of three additional rare plants - an historical sighting (not seen in the last 25 years) of Tuckerman's quillwort (*Isoetes tuckermanii*); a 1989 sighting of Robbin's spikerush (*Eleocharis robbinsii*) along the east shore of Little Lake and in the channel between Lake St. Catherine and Little Lake; and a 1973 sighting of whorled watermilfoil (*Myriophyllum verticillatum*) near the Fish and Wildlife public boat launch.

Non-chemical controls for watermilfoil have been used in Lake St. Catherine, Lily Pond and Little Lake including bottom barrier, mechanical harvesting, hydroraking and handpulling conducted by the Applicant, individuals with residences on the lakeshore, and/or private contractors. The efforts were conducted using private and municipal funds and funds derived from the Department's grant-in-aid program.

The Applicant and Co-Applicant propose a five-year plan combining the use of chemical and non-chemical methods to manage watermilfoil in Lake St. Catherine, Lily Pond and Little Lake. The chemical component of the plan involves the use of Sonar A.S. in a whole-lake treatment in year one to reduce the watermilfoil population, followed if needed in years three or four by either a second whole-lake chemical treatment using Sonar A.S. or a spot or partial-lake chemical treatment using Sonar A.S., Sonar P.R., Navigate/Aqua-Kleen or Renovate 3. Although a second whole-lake treatment is mentioned in the plan, the plan cost estimate breakdown underestimates the cost of this task. The non-chemical components of the plan include handpulling and bottom barrier, proposed in all five years, and regular, ongoing watermilfoil searches, education, and volunteer training throughout the five years. Mechanical harvesting and hydroraking are proposed in years three and four as optional strategies to chemical treatment only.

A. Aquatic Herbicide Description

Sonar A.S. is a U.S. Environmental Protection Agency-registered aquatic pesticide (EPA Registration No. 67690-4). Manufactured as an aqueous solution by SePRO Corporation, the active ingredient in Sonar A.S. is fluridone (1-methyl-3-phenyl-5-[3-(trifluoromethyl)phenyl]-4(1H)-pyridinone). Fluridone comprises 41.7 % of the formulation. Inert ingredients comprise the remaining 58.3 % of the formulation. The primary inert ingredient, water, comprises approximately 44.8 % of the formulation. The remaining eight inerts, comprising approximately 13.5 % of the formulation, are classified as proprietary information. These eight inerts are known to the Vermont Agency of Agriculture, Food and Markets, and the Vermont Department of Health.

Sonar A.S. is a selective systemic aquatic herbicide for management of aquatic vegetation in fresh water ponds, lakes, reservoirs, and drainage and irrigation canals. Plants take up the herbicide's active ingredient, fluridone, from the water through foliage and from sediments by roots, and translocate it throughout their systems. Fluridone kills susceptible plants by interfering with the plant's ability to make carotene, a plant pigment important for photosynthesis. Without the ability to make food, the plant dies. Susceptible plants become pinkish or whitish, and typically die within

30 to 90 days. The effectiveness of a Sonar A.S. treatment is a function of maintaining an adequate concentration of fluridone for a sufficient exposure time.

In an aquatic environment, fluridone is broken down by sunlight (the main degradation mechanism) and metabolically by plants and microorganisms. The half-life of fluridone and its breakdown products in a lake environment depends on the relative contribution of the different transport and transformation processes of hydrolysis, photolysis, volatilization, sorption, and microbial degradation acting in the environment. These processes can be influenced by water temperature, water pH, water depth, water clarity, turbidity, and photoperiod.

B. Proposed Chemical Treatment Plan

The Applicant and Co-Applicant propose a whole-lake chemical treatment with Sonar A.S. for year one of a five-year watermilfoil management plan. An initial application will consist of adding enough Sonar A.S. to the lakes to achieve a target in-lake concentration of 8 parts per billion (ppb) fluridone. More Sonar A.S. may be added (e.g., booster treatments) when the fluridone concentration drops by 2 to 4 ppb from the target concentration of 8 ppb. Fluridone concentration monitoring and the condition of the watermilfoil plants will dictate booster treatments. The Applicant and Co-Applicant propose to maintain a concentration of 4-8 ppb of fluridone in the three lakes for a period of 45 days. The Applicant and Co-Applicant anticipate that one or two booster treatments will be needed. However, due to factors that could influence the rate of decline of fluridone levels in the water, such as heavy rainfall (dilution) or high levels of sunshine (increased photodegradation), up to four booster treatments are proposed. The timing of the booster treatments is estimated at 14-day to 21-day intervals.

The Applicant and Co-Applicant expect the treatment program to begin between early May and early June, shortly after the establishment of a thermocline in Lake St. Catherine. The exact date of the initial treatment will depend on several factors such as watermilfoil growth, weather conditions and the necessary establishment of a thermocline in Lake St. Catherine. A thermocline will prevent Sonar A.S. from mixing into deeper, colder water and means less product will be needed to achieve the target concentration in the upper water column where the watermilfoil plants are growing. Lily Pond and Little Lake, both with an average depth of 4 feet, do not develop a thermocline. The 2001 report prepared by the Co-Applicant and ReMetrix included bathymetric information and generated water volumes for each four-foot change in water depth for the entire three-lake system. The three-lake system was divided into eight sections and the water volumes for each section determined. Accurate water volume information in conjunction with water temperature profiles will allow for precise calculations to determine the amount of Sonar A.S. to apply to each of the eight sections.

The most recent literature and manufacturer information document the importance of exposing watermilfoil to Sonar A.S. early in the growing season when the plant's growth resources have been weakened during the winter. If the initial treatment is conducted in mid to late June, watermilfoil may already be actively growing, at the surface of the lakes and forming dense mats. The Department therefore finds that the proposed initial treatment should not take place after June 15.

Based on previous experience in Vermont with whole-lake low-dose Sonar A.S. treatments, it is possible that more than four boosters could be needed for treatment success. The recent literature and manufacturer information indicate the need to maintain fluridone concentrations above 2 ppb for more than 60 and up to 90 days for "optimal" control of watermilfoil. However, recent Sonar A.S. treatments in Vermont, which included follow-up non-chemical control efforts, indicate the need to maintain concentrations above 4 ppb for at least 90 days in order to obtain at least two years of watermilfoil control. Longer than two years of control is desirable to minimize pesticide use and provide a more stable and diverse plant community. Therefore, the Department finds that maintaining a minimum fluridone concentration for a much longer period than the 45-day period following the initial treatment proposed by the Applicant and Co-Applicant is necessary to ensure an adequate fluridone concentration-exposure time to control watermilfoil for longer than two years. The Department intends to require a sustained fluridone concentration of at least 5 ppb for a minimum of 90 days.

The need for booster treatments will be based on water testing by the FasTEST method, an enzyme-linked immunosorbant assay test for detecting fluridone with an accuracy level of 1 ppb, to determine the level of fluridone within the waterbodies. The Applicant will be trained in FasTEST sample collection by SePRO Corporation and will collect the FasTEST samples. Collected samples will be shipped directly to SePRO Corporation for analysis. The Applicant and Co-Applicant propose to sample 12 sites in Lake St. Catherine, Lily Pond and Little Lake once per week for 12 weeks following each application of Sonar A.S. As Sonar A.S. may be detectable in the outlet stream, the Department intends to require sample sites in Mill Brook at locations approximately one-quarter mile and one mile downstream from the Little Lake outlet. In addition, since a sustained fluridone concentration of at least 5 ppb will be required to be maintained for a minimum of 90 days, the Department intends to require that weekly sampling continue for at least 90 days after the initial treatment. Weekly sampling will need to continue after 90 days if the fluridone concentration is 5 ppb or above at any location, until the fluridone concentration falls below 5 ppb at all locations, to determine the actual length of time that the fluridone concentration was at least 5 ppb and to determine when the use restriction for irrigation can be lifted.

The Sonar A.S. treatments will be performed by Vermont-licensed aquatic applicators. A subsurface application of Sonar A.S. is proposed using specially designed, calibrated, boat-mounted equipment that injects the herbicide underwater via weighted, trailing hoses. The chemical injection system dilutes concentrated liquid Sonar A.S. prior to its injection into the lake; the target concentration is achieved as the injected diluted herbicide mixes with lake water. A steady state concentration of Sonar A.S. generally occurs within two to five days of application. Until a steady state concentration is reached, fluridone levels may be higher than the target concentration. The injected solution is typically 1 part fluridone to approximately 121 parts water (or one gallon of Sonar A.S. to 50 gallons of water). A subsurface application reduces volatilization and allows for more even distribution of the chemical in the lake. Two spray boats with crews working simultaneously are anticipated to treat the three-lake system. The Applicant and Co-Applicant expect each treatment to be completed in one extended workday or possibly two days.

The current label for Sonar A.S. specifies the following water use restrictions for the use of Sonar A.S. in treated waters under a section entitled "General Use Precautions:"

- Potable Water Intakes: In lakes and reservoirs or other sources of potable water, DO NOT APPLY Sonar A.S. at application rates greater than 20 ppb within one-fourth mile (1320 feet) of any functioning potable water intake. At application rates of 6 - 20 ppb, Sonar A.S. MAY BE APPLIED where functioning potable water intakes are present. Note: Existing potable water intakes that are no longer in use, such as those replaced by potable water wells or connections to a municipal water system, are not considered to be functioning potable water intakes.
- Chemigation: Do not apply Sonar A.S. through any type of irrigation system.

- Hydroponic Farming: Do not use Sonar A.S. treated water for hydroponic farming.
- Greenhouse and Nursery Plants: Do not use Sonar A.S. treated water for irrigating greenhouse or nursery plants. Use of an approved assay should confirm that residues are <1 ppb (less than 1 ppb).
- Irrigation: Irrigation from a Sonar A.S. treated area may result in injury to the irrigated vegetation. SePRO Corporation recommends following the precautions and informing those who irrigate from areas treated with Sonar A.S. of the following irrigation time frames or water assay requirements in lakes and reservoirs where one-half or greater of the body of water is treated: 7 days after application for established tree crops, 30 days after application for established row crops/turf/plants, and water assay required prior to irrigation of newly seeded crops/seedbeds or areas to be planted including over-seeded golf course greens. These time frames and assay recommendations are suggestions which should be followed to reduce the potential for injury to vegetation irrigated with water treated with Sonar A.S. Greater potential for crop injury occurs where Sonar A.S. treated water is applied to crops grown on low organic and sandy soils.

  Where the use of Sonar A.S. treated water is desired for irrigating crops prior to the time frames established above, the use of FasTEST is recommended to measure the concentration in the treated water. Where FasTEST has determined that the concentrations are less than 10 ppb, there are no irrigation precautions for irrigating established tree crops, established row crops or turf. For tobacco, tomatoes, peppers or other plants within the Solanaceae family and newly seeded crops or newly seeded grasses such as over-seeded golf course greens, do not use Sonar A.S. treated water if measured fluridone concentrations are greater than 5 ppb. Furthermore, when rotating crops, do not plant members of the Solanaceae family in land that has been previously irrigated with fluridone concentrations in excess of 5 ppb. It is recommended that an aquatic specialist be consulted prior to commencing irrigation of these sites.

There are no municipal wells or water services provided by the towns of Poultney or Wells. All homes are supplied by either private wells or direct water intakes. There are no known uses of Mill Brook downstream of the Little Lake outlet. There are domestic and irrigation uses of Lake St. Catherine, Lily Pond and Little Lake. For the purposes of reviewing this project, the Department made the very conservative assumption that shoreland residents drink the lake water. There is a state park on the northeast shoreline of Lake St. Catherine and a public boat access area on the southwest shoreline.

The Applicant proposes that at a minimum, the temporary water use restrictions as specified on the current label will be followed. The Applicant recognizes that additional restrictions may be applied in the conditions of a permit, restrictions that could include the use of the outlet stream in addition to the use of the lakes. The Applicant is prepared to carry out these restrictions as required.

3. No Reasonable Non-chemical Alternative

Watermilfoil was discovered in Lake St. Catherine, Lily Pond and Little Lake in 1983. Mechanical harvesting and hydroraking have been the primary management methods used to control watermilfoil in the lakes. Mechanical harvesting has been conducted on the lakes annually since 1985. The Applicant currently owns and operates three harvesters during the summer season, authorized under Aquatic Nuisance Control Permit #2000-H03. Hydroraking was performed from

1996 through 2000 at a number of sites. To a lesser extent, bottom barriers and removing watermilfoil plants by hand have also been used.

In spite of non-chemical control efforts, the watermilfoil population in Lake St. Catherine, some areas of Lily Pond, and Little Lake continues to grow in dense surfacing mats in both shallow and deep water. At the present level of infestation, watermilfoil severely interferes with the public's use of the lakes for such recreational pursuits as swimming, boating and fishing.

A. Potential Alternatives

Before an Aquatic Nuisance Control Permit can be issued authorizing the use of a chemical pesticide under 10 V.S.A. § 1263a, the Applicant must demonstrate and the Secretary must find that there are no reasonable non-chemical alternatives available. Based on the Department's own work on Lake St. Catherine, Lily Pond and Little Lake and the 39 other lakes around the state where non-chemical methods have been used, and based on the information submitted by the Applicant and Co-Applicant, the Department does not know of a reasonable non-chemical alternative available for use in Lake St. Catherine, Lily Pond or Little Lake which would be effective at reducing watermilfoil growth to a level where recreational uses are no longer impaired for a period of several years or more. All known non-chemical alternatives have significant drawbacks which prevent them from being acceptable as the primary control method(s), either alone or in combination, to significantly reduce watermilfoil growth in Lake St. Catherine, Lily Pond and Little Lake.

- Installation of bottom barrier and associated barrier maintenance for an area the size of that proposed for chemical treatment would be extremely labor intensive and expensive. Based on actual costs incurred for bottom barrier installation to control one acre of watermilfoil at Lake Morey in Fairlee several years ago, it would cost several million dollars to treat the proposed areas in Lake St. Catherine, Lily Pond and Little Lake with bottom barrier instead of chemicals (not including annual maintenance costs). In addition, bottom barriers are not selective for watermilfoil. All plant species beneath the barriers would be killed, and the barriers would have significant adverse effects on benthic organisms. The use of bottom barrier on this scale would cause widespread destruction of aquatic habitat and pose an unacceptable risk to non-target organisms, making the use of this method infeasible.
- Diver-operated suction harvesting is primarily designed for small infestations because it is slow and labor intensive due to manual removal of the plants by SCUBA divers. Use of many suction harvesters at one time in an attempt to control a large watermilfoil infestation, such as that which occurs in Lake St. Catherine, Lily Pond and Little Lake, would pose an unacceptable risk to non-target organisms because of increased turbidity. Resettling of suspended sediment particles can pose a threat to fish eggs and aquatic invertebrates by smothering them. Increased turbidity also directly affects SCUBA diver visibility and would make it difficult for divers to effectively remove watermilfoil plants on this scale. A silt/fragment curtain enclosing the entire suction harvesting area in an effort to contain the turbidity would still pose an unacceptable risk to non-target organisms within the contained area, particularly since such a large area of the three-lake system would need to be suction harvested. Non-target plants would be impacted since it would be impossible for divers to selectively remove watermilfoil under these conditions. The cost of such an operation would also be prohibitive.

- Handpulling is slower and more labor intensive than suction harvesting, and would have the same negative impacts due to the number of handpullers that would be needed to control the area targeted for chemical treatment.
- Although mechanical harvesting has been used for watermilfoil control on Lake St. Catherine, Lily Pond and Little Lake since 1985 in an effort to provide immediate relief from dense watermilfoil growth, this method has not provided a satisfactory level of control, which is why the Applicant is now requesting to use an aquatic herbicide. Hydroraking and rotavating on the scale that would be necessary to control watermilfoil throughout these three waterbodies would have significant impacts to non-target organisms, including native plants and macroinvertebrates. The hydroraking that was conducted in 1996 - 2000 was approved for only a limited number of sites.
- Drawdowns of Lake St. Catherine, Lily Pond and Little Lake are not an option because there is not an existing outlet structure at Little Lake that would enable a significant lowering of the lakes. Even if a significant lowering of the lakes could be achieved the potential for negative impacts to area wetlands make this non-chemical method inappropriate. Drawdowns are not selective for watermilfoil and they can have severe negative impacts on many types of native plants that are important for fish and wildlife habitat, as well as having negative impacts on other aquatic biota.
- Grass carp, a plant-eating fish native to China, would not be appropriate for Lake St. Catherine, Lily Pond or Little Lake because beneficial native vegetation would be removed first and eventual removal of all vegetation would be likely, causing extensive negative impacts to the non-target environment. In addition, the introduction of grass carp is illegal in Vermont.
- Weevils have not yet proven to be effective in open water field settings where the insects have been intentionally introduced. No conclusive data is available at this time that documents that weevils can be used as a predictable and reliable watermilfoil control method. Although weevils already occur naturally in Lake St. Catherine, Lily Pond and Little Lake, they have not been successful at controlling the watermilfoil to date.

The Applicant proposes in the five-year integrated long-range management plan to use handpulling, installation of bottom barrier materials, spot/partial-lake treatment with herbicides (year three or four), and education and volunteer training after the whole-lake chemical application has reduced watermilfoil to the point where these methods are feasible.

The Department finds that the Applicant has met the statutory requirement to demonstrate that "... there is no reasonable non-chemical alternative available...."

4. Acceptable Risk to the Non-target Environment

Sonar A.S. is a systemic aquatic herbicide whose active ingredient, fluridone, kills susceptible species by interfering with the plant's ability to make food. Potential impacts to non-target organisms from the use of Sonar A.S. may be through direct toxic effects, or indirectly, through a physical change in habitat or shift in water quality conditions caused by the chemical that may affect some other component of the lake ecosystem.

A. Potential Direct Effects of Sonar A.S.

The aquatic plant communities in Lake St. Catherine, Lily Pond and Little Lake are still quite diverse in spite of dense watermilfoil growth. The 2001 report prepared by the Co-Applicant and ReMetrix identified 27 species of submerged, floating-leaved, and emergent plants in addition to watermilfoil. The Nongame and Natural Heritage Program of the Vermont Department of Fish and Wildlife informed the Department that there are records of three rare plants, an historical sighting of *Isoetes tuckermanii*; a record of *Eleocharis robbinsii* along the east shore of Little Lake and in the channel between Lake St. Catherine and Little Lake; and a record of *Myriophyllum verticillatum* near the Fish and Wildlife public boat launch. In addition to these records, a 1984 plant survey conducted by the Department identified the rare *Myriophyllum farwellii* in Lake St. Catherine. The Department also identified the rare *Potamogeton strictifolius* on site visits of Lake St. Catherine in 1989 and 1999, and the rare slender naiad *Najas gracillima* in Lake St. Catherine in 2003. The Nongame and Natural Heritage Program also has a record of an uncommon plant, *Utricularia gibba*, found on the eastern shore of Little Lake. Site visits conducted by the Department identified *Utricularia gibba* in 1990 and 2003 in Lake St. Catherine, and in 1998 in Lily Pond. Of the six rare and one uncommon plant species known from the three-lake system, only humped bladderwort, *Utricularia gibba*, was identified during the Co-Applicant and Re-Metrix survey conducted in 2001.

Of the 27 non-target plant species identified in Lake St. Catherine, Lily Pond and Little Lake by the Co-Applicant and Re-Metrix, the Applicant and Co-Applicant document eight aquatic plant species that are known to be susceptible, or intermediate to susceptible, to the aquatic herbicide Sonar A.S. at the proposed concentration of 8 ppb (*Brasenia schreberi*, *Ceratophyllum demersum, Elodea canadensis, Megalodonta beckii, Nuphar variegata, Nymphaea odorata, Potamogeton crispus,* and *Najas flexilis*). The Applicant and Co-Applicant expect significant impacts to these species in the year of treatment but expect recovery within one to two years of treatment. The Applicant and Co-Applicant also provided susceptibility information for the following rare or uncommon plants: *Eleocharis robbinsii, Isoetes tuckermanii*, *Myriophyllum verticillatum*, *Potamogeton strictifolius,* and *Utricularia gibba.* According to SePRO, *Eleocharis robbinsii* and *Isoetes tuckermanii* are tolerant to Sonar A.S. treatments, and *Potamogeton strictifolius* is inferred tolerant. *Utricularia gibba* is intermediate to tolerant to Sonar A.S. treatments. The potential impact to *Myriophyllum verticillatum* is uncertain, however, some species of watermilfoil have proven to be fairly tolerant of low concentrations of Sonar A.S. No information was provided regarding the susceptibility of the rare plants *Myriophyllum farwellii* and *Najas gracillima.*

Sonar, either as the aqueous solution (A.S.) or the slow-release pellet formulation (S.R.P.), has been used in at least 45 states for aquatic plant control and in at least 40 states for control of watermilfoil. The label allows for target treatment concentrations as high as 150 ppb to control a wide variety of plants. However, recent studies conducted in Michigan lakes have shown that when very low concentrations of Sonar are used (approximately 6 ppb), the herbicide can be quite selective, removing most of the watermilfoil (up to a 95% reduction) while causing no significant direct impacts to non-target aquatic and wetland plants, even in the year of treatment.

Four whole-lake low-dose treatments with Sonar A.S. have occurred to date in Vermont; treatments occurred in Burr Pond and Lake Hortonia in 2000, Sunrise Lake in 2001 and Beebe Pond in 2003. Of the treatments that have occurred in Vermont, the first three lakes were treated at a target concentration of 6 ppb with booster treatments that maintained the concentration above 4 ppb for at least 45-90 days. Beebe Pond was treated at a target rate of 8 ppb with booster treatments that maintained the concentration above 5 ppb for 90 days.

The Michigan studies documented that low concentrations of Sonar applied during whole-lake treatments had no negative effect on plant species diversity. In fact, plant diversity actually increased post-treatment after the competitive watermilfoil was reduced. Sonar treatments in

Michigan in 1998 also did not reduce overall native plant cover, which was maintained at levels greater than 75%. Two plant species that are susceptible to Sonar, *Ceratophyllum demersum* and *Elodea canadensis*, were not affected in the Michigan studies at the lower application rates.

Studies associated with the treatments in Vermont documented that whole-lake low-dose treatments had no major impact on overall abundance of native aquatic plants; species richness values on a whole-lake basis remained unchanged, as did native plant biomass levels.

There is potential for *Brasenia schreberi*, *Ceratophyllum demersum, Elodea canadensis, Megalodonta beckii*, *Nymphaea odorata, Nuphar variegata, Potamogeton crispus, Najas flexilis, and Najas gracillima* to be negatively impacted by a treatment. *Ceratophyllum demersum, Elodea canadensis* and *Nuphar variegata* were found in all four Vermont lakes previously treated with Sonar A.S. *Brasenia schreberi* was found in Sunrise Lake. *Megalodonta beckii* was found in Burr Pond, Lake Hortonia and Beebe Pond. *Nymphaea odorata* was found in Burr Pond, Lake Hortonia and Sunrise Lake. Based on information from these lakes, recovery of *Brasenia schreberi*, *Elodea canadensis, Megalodonta beckii*, *Nymphaea odorata* and *Nuphar variegata* is expected within one to two years following treatment. The recovery of *Ceratophyllum demersum* may take longer, but this species has been documented post-treatment in all four lakes.

Najas flexilis and *Najas gracillima* will likely be reduced in the year of treatment but should also recover in the year following treatment. Plants in the genus *Najas* are annuals that produce a heavy seed set each year. These seeds are resistant to the effects of fluridone and as a result, *Najas* species tend to increase in abundance the year following a Sonar treatment, presumably re-growing from the seed bank once the competitive non-native watermilfoil is removed. Data collected on *Najas* in association with the low-dose Sonar treatments in Vermont have shown significant increases in this genus in the year following treatment.

Like watermilfoil, *Potamogeton crispus* is nonnative. Due to its growth habit, *P. crispus* will typically cause nuisance conditions in the early part of the growing season (June and July), not throughout the season like watermilfoil. Although this species may be negatively impacted by a Sonar A.S. treatment, data collected on *P. crispus* in association with the low-dose Sonar treatment in Lake Hortonia have shown significant increases in this species in the year following treatment.

If present in the three-lake system, there is the potential for *Myriophyllum verticillatum* and *M. farwellii* to be impacted by an 8 ppb Sonar A.S. treatment, both positively and negatively. While the impact to these species is uncertain, some native watermilfoil species have proven to be fairly tolerant of low concentrations of fluridone (*M. humile*) or slightly more tolerant (*M. sibericum*) of fluridone than the nonnative watermilfoil (*M. spicatum*) appears to be. *M. sibericum* recovered from low-dose treatments in Vermont (Burr Pond and Lake Hortonia) to pre-treatment levels within two years of treatment and in Michigan, within one year after treatment. This is perhaps a result of decreased competition from the nonnative watermilfoil.

The above information, along with field observations of treatments conducted in other states over the last few years, suggests that Sonar A.S. may be used as described in the Applicant's and Co-Applicant's proposal with little to no adverse effect on beneficial native plant species within the year of treatment, including such species as *Vallisneria americana*, *Chara* sp., *Typha* sp., many *Potamogeton* spp., and *Zosterella dubia,* all found in Lake St. Catherine, Lily Pond and/or Little Lake. In fact, many of these Sonar-tolerant submerged plants are likely to increase in abundance, possibly even within the year of treatment, as the watermilfoil dies. If *Brasenia schreberi*, *Ceratophyllum demersum*, *Elodea canadensis*, *Megalodonta beckii*, *Nuphar variegata*, *Nymphaea odorata*, *Potamogeton crispus,* and/or *Najas flexilis* are negatively impacted by the treatment, the population reductions are not expected to last beyond one to two years post-treatment. Collection of specific information on the abundance of these eight susceptible species will be required in pre- and post-treatment aquatic plant surveys. Specific information on the six rare and one uncommon species known from the three-lake system will also be required in the pre- and post-treatment aquatic plant surveys.

Lake St. Catherine, Lily Pond and Little Lake are mapped as lacustrine palustrine systems on the Department's wetland inventory maps. Class II wetlands exist 1) at the north and south ends of Lily Pond as scrub shrub and emergent wetlands, 2) along the inlet to Little Lake from Lake St. Catherine as scrub shrub and emergent wetlands, and 3) at the south end of Little Lake as scrub shrub and forested wetlands. The Department's Wetlands Office recommended that the Applicant monitor the aquatic vegetation within the open water areas in the Class II wetlands on Lake St. Catherine, Lily Pond and Little Lake as described above for the duration of the five-year integrated management plan to determine whether the treatment was successful for watermilfoil control as well as to assess potential impacts to non-target plants.

Sonar A.S. is not directly toxic to aquatic organisms such as fish, waterfowl, and invertebrates when used at the rates recommended on the product label. A wide margin of safety has been indicated in field and laboratory toxicity tests and there have been no reports from other states where Sonar is used, even at the maximum label rate, that it has had a direct toxic effect on any aquatic organisms.

An October 1999 summary of research on the effects of Sonar on aquatic organisms (*Evaluation of the Use of Sonar in Michigan* by the Michigan Environmental Science Board) reported that Sonar, when applied according to the label, does not accumulate in fish; does not increase mortality, decrease body condition, or modify behavior in carp; is well below lethal levels to fish; has no observable adverse effects on growth or survival of larval fish; shows no effects (accumulation, or acute or chronic toxicity) on benthic or epiphytic macroinvertebrates; and shows no acute or chronic toxicity to zooplankton. A herpetological study conducted for the Vermont Department of Fish and Wildlife in conjunction with the 2001 Sonar A.S. treatment of Sunrise Lake found no obvious or alarming declines in amphibian or reptile species in Sunrise Lake in the year of treatment and one year post-treatment. A final report on the study is not yet available.

A review of the potential effects on fish and macroinvertebrates of Sonar A.S. (fluridone), its major metabolites, and its inert ingredients was conducted by biologists in the Department's Biomonitoring and Aquatic Studies Section. The inert ingredients were reviewed through toxicity information provided by the Vermont Agency of Agriculture, Food and Markets without specifically identifying the inert ingredients. It was determined that treatment at a target concentration of 8 ppb should pose an acceptable risk to the fish and invertebrate communities in all three lakes.

Based on the above information, the Department does not intend to require the Applicant to inventory the fauna of the Class II wetlands either pre- or post-treatment.

Fisheries and wildlife biologists within the Vermont Department of Fish and Wildlife also conducted a review of the proposed project. They indicated the importance of requiring careful monitoring of Sonar concentrations in the water column to prevent over-treatment, which could result in some unanticipated impacts to non-target aquatic and wetland vegetation. Vermont Department of Fish and Wildlife staff recommended that monitoring of the aquatic vegetation in Lake St. Catherine, Lily Pond and Little Lake be conducted for the duration of the five-year integrated management plan to determine whether the treatment was successful for watermilfoil control as well as to assess potential impacts to non-target plants.

A Department of Fish and Wildlife Nongame and Natural Heritage Program botanist recommended that a fluridone target rate of no greater than 6 ppb be permitted. The botanist expressed concern that if the requested concentration of 8 ppb were permitted, a concentration closer to 10 ppb might result. However, the Department finds that following application, Sonar A.S. will mix to the thermocline level, if a thermocline is established. Water column temperature profile measurements collected just prior to treatment linked with accurate lake bathymetry will allow for precise application of the target concentration to the water above the thermocline. While the concentration of fluridone may exceed the target concentration until it is completely mixed throughout the treatment area, elevated concentrations should only last for a few days and not for a time period that would cause a risk to non-target aquatic plants beyond the risk already described above.

The agricultural toxicologist at the Vermont Agency of Agriculture, Food and Markets informed the Department that none of the nine inert ingredients in Sonar A.S. are found on EPA's List 1, "Inerts of Toxicological Concern," or List 2, "Potentially Toxic Inerts with High Priority for Testing."

Sonar A.S. may have a direct toxic effect on some terrestrial crop plants. The current label for Sonar A.S. has an irrigation precaution which recommends that when one-half or greater of the area of a lake or reservoir is treated, the water should not be used for 7 days on established tree crops, and for 30 days on row crops, turf and plants. In addition, a water assay analysis is required prior to irrigation use for terrestrial crop plants in the Solanaceae family, such as tomatoes, peppers and tobacco, and for newly seeded crops, seedbeds, or newly seeded grasses including over-seeded golf course greens, to assure that the concentration of fluridone in the irrigation water does not exceed 5 ppb. If these precautions are followed it is unlikely that there will be toxic effects on terrestrial plants.

Dense watermilfoil beds, particularly those that cover a high percentage of a lake's surface area or littoral zone, have the potential to cause many changes in the lake environment, which can both directly and indirectly impact aquatic organisms. Some of these impacts include reduced oxygen levels; a significant increase in water temperature; changes in lake nutrient dynamics and sediment loading; displacement of native and/or endangered, threatened or rare aquatic plant species; changes in fish spawning site availability; changes in horizontal and vertical fish distribution; and reduction in feeding success of predatory fish.

Vermont Sonar A.S. treatments have been deemed successful at reducing levels of watermilfoil in the year of treatment by at least 90% in Burr Pond and Lake Hortonia, and an estimated 99% in Sunrise Lake, while causing no significant direct impacts to the majority of non-target aquatic and wetland plants, even in the year of treatment. However, by year two post-treatment, watermilfoil had reinfested Lake Hortonia and Burr Pond. In Sunrise Lake, a relatively small lake with characteristics conducive to successful control through intensive handpulling, watermilfoil regrowth was significant as well, but ongoing handpulling and bottom barrier follow-up measures may be effectively controlling it. It is still too soon to evaluate the long-term success of the follow-up non-chemical control measures on Sunrise Lake. Beebe Pond was treated in 2003; it is too soon to evaluate the success of this treatment and any follow-up non-chemical control measures.

Based on the results of the Vermont Sonar A.S. treatments to date, treatments with Sonar A.S. at 6 ppb that maintain concentrations above 4 ppb for 45-90 days appear to provide relatively short-lived control (1-2 years post-treatment). To provide a more stable and diverse native plant community, longer-term control is desirable. The Department believes the use of Sonar A.S. at 8 ppb as proposed by the Applicant and Co-Applicant may provide longer-term watermilfoil control while posing an acceptable risk to the non-target environment if at least 5 ppb of fluridone is maintained for a minimum of 90 days.

Overall, the Department finds that considering the potential impacts of a Sonar A.S. treatment described above, as well as the continued impacts in Lake St. Catherine, Lily Pond and Little Lake of a widespread watermilfoil population, the direct impacts from a Sonar A.S. treatment pose an acceptable risk to the non-target community of Lake St. Catherine, Lily Pond and Little Lake.

B. Potential Indirect Effects of Sonar A.S.

Indirect impacts to non-target organisms such as fish, waterfowl, and macroinvertebrates can occur from the use of an aquatic herbicide if the product used is not selective for the target plant or if the target plant growth is so extensive that it comprises a significant portion of the habitat in the lake. Extensive vegetation removal results in loss of substrate, cover, and food for these organisms. This situation is not expected to occur in Lake St. Catherine, Lily Pond or Little Lake because Sonar A.S. is relatively selective at low concentrations. A diverse native plant community exists in these lakes, and many native plant species should remain to provide essential habitat after treatment occurs.

When fast-acting herbicides are used in lakes, there is potential for aquatic organisms to be impacted indirectly due to temporarily depressed oxygen levels caused by rapidly decomposing aquatic plants. Sonar A.S. works slowly, over a period of 30 to 90 days, making significant oxygen depletion unlikely. In addition, the treatment is proposed for spring when plant biomass is lower, further reducing the risk of depressed oxygen levels. Another potential impact of herbicide treatments can be the release of the nutrient phosphorus from decomposing vegetation. It is quite possible that an algae bloom, caused by increased phosphorus levels, may occur as a result of the Sonar A.S. treatment in Lake St. Catherine, Lily Pond and Little Lake. However, this increase in algae abundance would be temporary because the released phosphorus would eventually be bound in the bottom sediments, taken up by native rooted aquatic vegetation, or flushed from the lakes.

The Department finds that the chemical treatment of Lake St. Catherine, Lily Pond and Little Lake with Sonar A.S. at 8 ppb poses an acceptable risk to the non-target environment if it is conducted in accordance with the product label, the submitted proposal, and the conditions of this permit.

5. Negligible Risk to Public Health

The Vermont Department of Health has reviewed the proposed project to use Sonar A.S. in a whole-lake treatment of Lake St. Catherine, Lily Pond and Little Lake. The Department of Health has examined the potential level of concern for public health that may be associated with exposure to water that has been treated with Sonar A.S. and has made the following finding:

> Although the federal product label for Sonar A.S. indicates that no potable water use restrictions are necessary when whole-lake or reservoir treatments are conducted to achieve less than or equal to 20 ppb fluridone, due to the influence of many site-specific factors, it cannot indicate what the maximum concentration of active ingredient may be in the waters of concern at any location at any specific time post application.

Therefore, if Sonar A.S. is to be used as proposed in a whole-lake treatment of Lake St. Catherine, Lily Pond and Little Lake, the Department of Health recommends that certain water use restrictions beyond the federal label requirements be instituted in order to ensure protection of public health.

The following recommended water use conditions are based upon Department of Health review of the most current scientific information available for fluridone including any potential health effects, the half-life of the compound, consideration of who is likely to come into contact with treated waters and in what manner, several very health protective assumptions, standard risk assessment procedures and the assumption that only Sonar A.S. will be used to treat the waterbodies in question. In addition, based on a review by the State Toxicologist for the Department of Health, it is reasonable to conclude that human exposure to the inert compounds contained in Sonar A.S., as well as to any potential fluridone metabolites, is not likely to result in an increase in the level of concern for public health at the concentrations that are likely to result under the following conditions.

Specific Recommendations

No use of Lake St. Catherine, Lily Pond and Little Lake and associated outlet stream(s) (for one mile downstream of the effluent) for any purpose (including recreational uses such as boating and fishing) is recommended on the day of application. Boating, fishing and toilet flushing may resume at the beginning of the day following application. Swimming may resume 24 hours after application. However, domestic use (other than toilet flushing) should not resume until the conditions that follow have been met.

Twenty-four hours after the initial application of Sonar A.S., representative samples of the treated waterbodies and outlet stream(s) (within one-quarter mile of the effluent) should be chemically tested to determine if fluridone is present at less than or equal to 20 ppb. In addition, because there is some question as to whether n-methyl formamide (NMF)[1] may be formed in waters that have been treated with fluridone, these samples should also be chemically analyzed to determine if NMF is present using a detection limit of no more than 2 ppb. Analysis of multiple samples for both compounds is necessary in order to account for the influence of many chemical, media and site-specific factors.

If fluridone residues are confirmed to be less than or equal to 20 ppb and NMF is not detected, full use of Lake St. Catherine, Lily Pond, Little Lake, the outlet stream(s) and their waters, including all domestic uses, may resume. However, if fluridone is detected in representative samples from these waters in excess of 20 ppb and/or NMF is detected, an additional 24-hour waiting period should occur during which time Lake

1 In a controlled laboratory study, n-methyl formamide (NMF), a potential toxin, was reported to be one of several potential photodegradation by-products of fluridone. NMF has never been detected in Sonar-treated water outside of the laboratory under natural conditions and is not expected to be found in Vermont.

St. Catherine, Lily Pond, Little Lake and the outlet stream(s) (within one mile downstream of the effluent) should again not be used as a domestic water supply (except for toilet flushing). At the end of this second 24-hour waiting period, representative samples of Lake St. Catherine, Lily Pond, Little Lake and the outlet stream(s) (within one-quarter mile downstream) should again be taken and chemically analyzed for fluridone and/or NMF depending upon which condition(s) has not been met. This process should be repeated until representative sampling indicates that the level of fluridone in Lake St. Catherine, Lily Pond, Little Lake and the outlet stream(s) is less than or equal to 20 ppb and no NMF is detectable. Furthermore, this process applies to <u>any and all</u> booster applications.

Only once fluridone residues are confirmed to be less than or equal to 20 ppb <u>and</u> NMF is not detected, should full use of Lake St. Catherine, Lily Pond, Little Lake and the outlet stream(s) resume. Until full use can be resumed, bottled water should be supplied by the Applicant to those who may depend upon Lake St. Catherine, Lily Pond, Little Lake and/or the outlet stream(s) (within one mile of the effluent) for their domestic water supply.

Public notification of property owners and residents of the Lake St. Catherine, Lily Pond and Little Lake area as well as commercial camps and parents whose children are attending camps which use Lake St. Catherine, Lily Pond, Little Lake and/or waters within one contiguous water-mile of Lake St. Catherine, Lily Pond and Little Lake will occur 30 days prior to application. Waterbody access areas as well as any nearby campgrounds should be posted.

Based on the photolysis and behavior of fluridone in aquatic systems, the concentration of fluridone resulting from a Sonar A.S. treatment with a target concentration of 8 ppb should not be sufficient to result in the formation of NMF above a detection limit of 2 ppb. Under a realistic worst-case scenario, a concentration of greater than 30 ppb of fluridone would be needed to form 2 ppb of NMF. Therefore, the Department has determined that water samples need to be analyzed for NMF only if a fluridone concentration at or above 30 ppb is found.

Based on the above information, the Department finds that the proposed project poses a negligible risk to public health if permit conditions are followed.

6. Long-range Management Plan

The integrated management plan (IMP) proposed by the Applicant combines the use of chemical and non-chemical methods over five years to manage the infestation of watermilfoil in Lake St. Catherine, Lily Pond and Little Lake. The goal of the five-year IMP is to reduce the watermilfoil population during the first year through whole-lake chemical treatment and use non-chemical methods and spot/partial-lake chemical treatment, if needed, in subsequent years. A major component of the Applicant's IMP is monitoring of the aquatic plant community post whole-lake treatment. Beginning in the year following whole-lake treatment, surveys are proposed early in the growing season to detect watermilfoil regrowth and determine appropriate management strategies for the upcoming summer months. A follow-up, more comprehensive survey will be conducted mid to late summer to provide quantitative documentation of the aquatic plant coverage in the three lakes.

The two primary non-chemical control strategies for scattered or small patches of watermilfoil growth will be bottom barriers and handpulling. Larger infestations, areas greater than 0.5 acres in size, will be evaluated for spot/partial-lake chemical treatments. Herbicides under consideration include: Sonar A.S. in conjunction with a suspended containment system; Sonar PR, a precision release pellet formulation of fluridone; Navigate or Aqua-Kleen, granular formulations containing the active ingredient, 2,4-D; or Renovate, a granular formulation containing the active ingredient, triclopyr. Spot/partial-lake chemical treatment is anticipated for use in year three or four of the IMP, or possibly another whole-lake Sonar A.S. treatment. If follow-up chemical treatment were not allowed, the Applicant would explore mechanical harvesting and hydroraking as optional strategies to provide short-term relief for lake users.

Which follow-up method will be required in the lakes cannot be determined until post whole-lake treatment watermilfoil assessments are made; however, the Applicant is prepared to submit the necessary permit applications to the Department (e.g., bottom barrier materials, spot/partial-lake chemical treatment, mechanical harvesting).

The Applicant also proposes educational and volunteer training efforts to increase awareness of the control program and recruit volunteers to assist with the IMP, to reduce the likelihood that the lakes will be reinfested with watermilfoil. The Applicant publishes a newsletter twice a year and will provide regular updates on the IMP in the newsletter. Signs will be posted at all boat launching stations and the feasibility of installing boat washing stations will be explored.

To effectively evaluate the best follow-up watermilfoil management strategy, the Department intends to require a meeting with the Applicant on an annual basis prior to initiation of each phase of the IMP.

The Applicant has outlined a preliminary budget for the IMP. Implementation of the five-year plan is estimated to cost between $308,520 and $448,160, depending on factors such as the number of Sonar A.S. booster applications needed, how effective the whole-lake treatment is, the extent to which non-chemical methods or spot/partial-lake chemical treatments are needed, etc. The first year of the IMP is estimated at $203,520 - $253,160. Annual costs for the non-chemical portion of the IMP in years two thru five are estimated to be $12,500 - $22,500/yr. Additional costs in years three or four of $60,000 - $120,000/yr are estimated if spot/partial-lake treatments are employed.

The Applicant and Co-Applicant have proposed to maintain a concentration of 4-8 ppb of fluridone in the three waterbodies for a period of 45 days. Based on the results of previous Sonar A.S. treatments in Vermont, the Department finds that a minimum sustained fluridone concentration of at least 5 ppb for a minimum of 90 days will be necessary to have a reasonable expectation that watermilfoil can be controlled by non-chemical methods for longer than two years post-treatment and long-term pesticide use can be minimized.

By employing all of the components identified above in an integrated fashion over five years, the Applicant is seeking to control watermilfoil in Lake St. Catherine, Lily Pond and Little Lake in order to restore recreational uses as well as enhance the native aquatic plant community and ensure habitat diversity. While the Applicant has only proposed a five-year plan, the Applicant recognizes that management of watermilfoil will be an annual undertaking that needs to continue well beyond the five years. A diligent and sustained effort in the years proposed in the IMP and beyond will be required to prevent Lake St. Catherine, Lily Pond and Little Lake from becoming reinfested to the point where recreational uses and the ecology of the lakes are threatened.

The Department finds that the Applicant has incorporated a schedule of pesticide minimization over the long term by developing a plan that reduces watermilfoil growth in a single season to a level where it can be effectively controlled by means other than continued whole-lake chemical use. The Department finds that the five-year IMP has a reasonable chance of achieving its goal provided that the Sonar A.S. treatment is conducted in accordance with the conditions of this permit. The Department recognizes that there is a potential for even the modified chemical treatment and the proposed non-chemical control efforts to be unsuccessful at managing the watermilfoil population. Any request by the Applicant to conduct a future chemical treatment of either a whole-lake treatment or spot/partial-lake treatments to control the watermilfoil population will be evaluated in light of the success of the first Sonar treatment and the intensity of the follow-up non-chemical control efforts conducted by the Applicant.

The Department would consider the IMP to be successful if at the end of the proposed five-year IMP, watermilfoil in Lake St. Catherine, Lily Pond and Little Lake remains manageable by efforts other than a whole-lake chemical treatment, and a strong framework exists for continuing the management efforts indefinitely.

7. Public Benefit

The Department has determined that the use of the aquatic herbicide, Sonar A.S., as part of an IMP combining chemical and non-chemical control technologies to manage watermilfoil in Lake St. Catherine, Lily Pond and Little Lake, will provide a public benefit. It will enhance the native aquatic plant community in the bodies of water by removing a significant portion of the invasive watermilfoil population, allowing native plant species to spread, thus restoring habitat diversity. The proposed project will also significantly improve the recreational use of the lakes, which is currently significantly impaired by watermilfoil. Removal of dense surfacing stands of watermilfoil will greatly improve opportunities for swimming, fishing and boating. Control of watermilfoil in Lake St. Catherine, Lily Pond and Little Lake will also help prevent watermilfoil fragments from being easily transported from these lakes to other bodies of water on boat motors and trailers.

The issuance of this permit may be appealed to the Vermont Water Resources Board, National Life Records Center Building, Drawer 20, Montpelier VT 05620-3201 (telephone 802-828-3309), within 30 days of the date of this permit pursuant to 10 V.S.A. § 1269.

Dated at Waterbury, Vermont this ____ day of __________2003

Jeffrey Wennberg, Commissioner
Department of Environmental Conservation

By____

Wallace McLean, Director
Water Quality Division

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Attachment A

Herbicide Application Record Form

1. Name and location (town) of lake(s) treated ____________________
2. Date of treatment____________________
3. Time of treatment ____________________
4. Product trade name and formulation of herbicide used ____________________
5. Product manufacturer ____________________
6. Objective(s) of herbicide treatment ____________________
7. Total amount of herbicide used (gallons, quarts, etc.) along with chemical treatment quantity calculations____________________
8. Date thermocline measured and the thermocline depth (m) along with water column temperature profile measurements used for herbicide amount calculation ____________________
   ____
9. Number of acres treated ____________________
10. Target concentration of herbicide in water column (ppb) along with target concentration calculations ____________________
11. Herbicide application technique ____________________
    ____
    ____
    ____
12. Equipment used ____________________
13. Amount of time required to complete herbicide application ____________________
14. Weather and lake conditions at the time of treatment (rain, wind, wave action)____________________
15. Describe procedures taken to dispose of surplus product, empty containers, and rinseate.
    ____
    ____
    ____
    ____
16. Problems encountered ____________________
    ____
    ____
    ____
17. Name of Company (Co-Permittee) conducting treatment ____________________
18. Name(s) of all company personnel on-site during treatment ____________________
    ____
19. Comments: ____________________

Signed:

Permittee ____________________ Co-Permittee ____________________

Dated ____________________ Dated ____________________

Attachment B

State of Vermont
Department of Environmental Conservation
Request for Co-Permittee Status

I hereby request authorization, on behalf of myself as an individual or for ____ (Company), to become a Co-Permittee to use Sonar A.S. as approved by issuance of Aquatic Nuisance Control (ANC) Permit #2001-C08 to control Eurasian watermilfoil in Lake St. Catherine, Lily Pond and Little Lake in Poultney and/or Wells, Vermont. I hereby certify that I have read and am familiar with the terms and conditions of the aforementioned permit and agree to comply with all permit conditions that pertain to the Co-Permittee and/or work conducted by the Co-Permittee.

Name of Permittee: ____

Signature and Title of Permittee's Authorized Representative:

____

Date: __________________

Name of Proposed Co-Permittee's Representative:

____

Company Name: _____________________________________

Address: _____________________________________

____

____

Business Phone/FAX: (_____)________ /(_____)________

Signature and Title of Proposed Co-Permittee's Representative:

____

Date: __________________

Submit request to: VT Department of Environmental Conservation
Water Quality Division
103 South Main Street, Building 10 North
Waterbury, VT 05671-0408