Triclopyr Butoxyethyl Ester
Analysis of Risks
to
Endangered and Threatened Salmon and Steelhead

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Michael Patterson, Ph.D.
Environmental Field Branch
Office of Pesticide Programs

Summary

Triclopyr is a selective herbicide used on rice, rangeland, and pasture, rights-of-way, forestry, and turf, including home lawns for control of broadleaf plants and woody plants. There are currently 12 registered products containing tricliphyr butoxyethyl ester (BEE) and 24 products containing triclopyr triethylamine salt (TEA). This review is selective and the opinions given are limited to BEE use in forest trees.

Triclopyr functions as an herbicide in forestry applications by controlling broadleaf weeds and woody, deciduous trees in conifer forests. It is approved for a wide range of noxious weeds and woody plants. A principal use of the agent is in commercial and government lands in the process of reforestation following harvest or in the aftermath of fire related tree loss. The agent can, however, under current labels be applied to any coniferous forest. Application times vary, but are generally early summer, after broadleaf plants have fully extended their leaves, or early fall as conifers enter the winter dormant period. The agent is effective only on actively growing plants.

The main use of triclopyr is to control encroaching woody plants in emerging conifer forests. BEE toxicity testing indicated that it is moderately to highly toxic to freshwater fish and estuarine/marine invertebrates, slightly to moderately toxic to freshwater invertebrates, and highly toxic to estuarine/marine fish. Under earlier label requirements the Levels of Concern (LOC) were exceeded for many applications and species. The current Registration Eligibility Decision (RED) includes significant reductions in maximum application rates and includes Risk Quotients (RQs) that are reduced for most species, except freshwater fish.

Several factors reduce the Agency's concern regarding BEE, including the use of several worst-case exposure assumptions that are unlikely under actual use conditions. The acute risk to fish is based on direct application to shallow, static water. Direct application to water is no longer allowed and, in the areas being reviewed for this report the water is characterized as fast flowing and frequently deep.

[1] **Comment: Data and the analysis based upon it reflects information available at the time this report was completed. Additional data, which may be submitted or change in status after the submission date are not included in the authors evaluations, presentations, or comments.**

Scope - Although this analysis is specific to 3 listed chinook, coho, and steelhead in California and southern Oregon and the watersheds in which they occur, it is acknowledged that triclopyr is registered for uses that may occur outside this geographic scope and that additional analyses may be required to address other T&E species in the Pacific states as well as across the United States. I understand that any subsequent analyses, requests for consultation, and resulting Biological Opinions may necessitate that Biological Opinions relative to this request be revisited, and could be modified. Much of the quantitative information presented and used was derived from the Registration Eligibility Decision (RED, Attachment 1).

Contents

Attachments:

1. Background

Under section 7 of the Endangered Species Act, the Office of Pesticide Programs (OPP) of the U. S. Environmental Protection Agency (EPA) is required to consult on actions that may affect Federally listed endangered or threatened species or that may adversely modify designated critical habitat. Situations where a pesticide may affect a fish, such as any of the salmonid species listed by the National Marine Fisheries Service (NMFS), include either direct or indirect effects on the fish. Direct effects result from exposure to a pesticide at levels that may cause harm.

Acute Toxicity - Relevant acute data are derived from standardized toxicity tests with lethality as the primary endpoint. These tests are conducted with what is generally accepted as the most sensitive life stage of fish, i.e., very young fish from 0.5-5 grams in weight, and with species that are usually among the most sensitive. These tests for pesticide registration include analysis of observable sublethal effects as well. The intent of acute tests is to statistically derive a median effect level; typically the effect is lethality in fish (LC50) or immobility in aquatic invertebrates (EC50). Typically, a standard fish acute test will include concentrations that cause no mortality, and often no observable sublethal effects, as well as concentrations that would cause 100% mortality. By looking at the effects at various test concentrations, a dose-response curve can be derived, and one can statistically predict the effects likely to occur at various pesticide concentrations; a well done test can even be extrapolated, with caution, to concentrations below those tested (or above the test concentrations if the highest concentration did not produce 100% mortality).

OPP typically uses qualitative descriptors to describe different levels of acute toxicity, the most likely kind of effect of modern pesticides (Table 1). These are widely used for comparative purposes, but must be associated with exposure before any conclusions can be drawn with respect to risk. Pesticides that are considered highly toxic or very highly toxic are required to have a label statement indicating that level of toxicity. The FIFRA regulations [40CFR158.490(a)] do not require calculating a specific LC50 or EC50 for pesticides that are practically non-toxic; the LC50 or EC50 would simply be expressed as >100 ppm. When no lethal or sublethal effects are observed at 100 ppm, OPP considers the pesticide will have "no effect" on the species.

Table 1. Qualitative descriptors for categories of fish and aquatic invertebrate toxicity (from Zucker, 1985)

LC50 or EC50	Category description
< 0.1 ppm	Very highly toxic
0.1- 1 ppm	Highly toxic
>1 < 10 ppm	Moderately toxic
> 10 < 100 ppm	Slightly toxic
> 100 ppm	Practically non-toxic

Comparative toxicology has demonstrated that various species of scaled fish generally have equivalent sensitivity, within an order of magnitude, to other species of scaled fish tested under the same conditions. Exceptions are known to occur for only an occasional pesticide, as based on the several dozen fish species that have been frequently tested. Sappington et al. (2001), Beyers et al. (1994) and Dwyer et al. (1999), among others, have shown that endangered and threatened fish tested to date are similarly sensitive, on an acute basis, to a variety of pesticides and other chemicals as are their non-endangered counterparts.

Chronic Toxicity - OPP evaluates the potential chronic effects of a pesticide on the basis of several types of tests. These tests are often required for registration, but not always. If a pesticide has essentially no acute toxicity at relevant concentrations, or if it degrades very rapidly in water, or if the nature of the use is such that the pesticide will not reach water, then chronic fish tests may not be required [40CFR158.490]. Chronic fish tests primarily evaluate the potential for reproductive effects and effects on the offspring. Other observed sublethal effects are also required to be reported. An abbreviated chronic test, the fish early-life stage test, is usually the first chronic test conducted and will indicate the likelihood of reproductive or chronic effects at relevant concentrations. If such effects are found, then a full fish life-cycle test will be conducted. If the nature of the chemical is such that reproductive effects are expected, the abbreviated test may be skipped in favor of the full life-cycle test. These chronic tests are designed to determine a “no observable effect level” (NOEL) and a “lowest observable effect level” (LOEL). A chronic risk requires not only chronic toxicity, but also chronic exposure, which can result from a chemical being persistent and resident in an environment (e.g., a pond) for a chronic period of time or from repeated applications that transport into any environment such that exposure would be considered “chronic”.

As with comparative toxicology efforts relative to sensitivity for acute effects, EPA, in conjunction with the U. S. Geological Survey, has a current effort to assess the comparative toxicology for chronic effects also. Preliminary information indicates, as with the acute data, that endangered and threatened fish are again of similar sensitivity to similar non-endangered species.

Metabolites and Degradates - Information must be reported to OPP regarding any pesticide metabolites or degradates that may pose a toxicological risk or that may persist in the environment [40CFR159.179]. Toxicity and/or persistence test data on such compounds may be required if, during the risk assessment, the nature of the metabolite or degradate and the amount that may occur in the environment raises a concern. If actual data or structure-activity analyses are not available, the requirement for testing is based upon best professional judgement.

Inert Ingredients - OPP does take into account the potential effects of what used to be termed “inert” ingredients, but which are beginning to be referred to as “other ingredients”. OPP has classified these ingredients into several categories. A few of these, such as nonylphenol, can no longer be used without including them on the label with a specific statement indicating the potential toxicity. Based upon our internal databases, I can find no product in which nonylphenol is now an ingredient. Many others, including such ingredients as clay, soybean oil, many polymers, and chlorophyll, have been evaluated through structure-activity analysis or data and determined to be of minimal or no toxicity. There exist also two additional lists, one for inerts with potential toxicity which are considered a testing priority, and one for inerts unlikely to be toxic, but which cannot yet be said to have negligible toxicity. Any new inert ingredients are required to undergo testing unless it can be demonstrated that testing is unnecessary.

The inerts efforts in OPP are oriented only towards toxicity at the present time, rather than risk. It should be noted, however, that very many of the inerts are in exceedingly small amounts in pesticide products. While some surfactants, solvents, and other ingredients may be present in fairly large amounts in various products, many are present only to a minor extent. These include such things as coloring agents, fragrances, and even the printers ink on water soluble bags of pesticides. Some of these could have moderate toxicity, yet still be of no consequence because of the negligible amounts present in a product. If a product contains inert ingredients in sufficient quantity to be of concern, relative to the toxicity of the active ingredient, OPP attempts to evaluate the potential effects of these inerts through data or structure-activity analysis, where necessary.

For a number of major pesticide products, testing has been conducted on the formulated end-use products that are used by the applicator. The results of fish toxicity tests with formulated products can be compared with the results of tests on the same species with the active ingredient only. A comparison of the results should indicate comparable sensitivity, relative to the percentage of active ingredient in the technical versus formulated product, if there is no extra activity due to the combination of inert ingredients. I note that the “comparable” sensitivity must take into account the natural variation in toxicity tests, which is up to 2-fold for the same species in the same laboratory under the same conditions, and which can be somewhat higher between different laboratories, especially when different stocks of test fish are used.

The comparison of formulated product and technical ingredient test results may not provide specific information on the individual inert ingredients, but rather is like a “black box” which sums up the effects of all ingredients. I consider this approach to be more appropriate than testing each individual inert and active ingredient because it incorporates any additivity, antagonism, and synergism effects that may occur and which might not be correctly evaluated from tests on the individual ingredients. I do note, however, that we do not have aquatic data on most formulated products, although we often have testing on one or perhaps two formulations of an active ingredient.

Risk - An analysis of toxicity, whether acute or chronic, lethal or sublethal, must be combined with an analysis of how much will be in the water, to determine risks to fish. Risk is a combination of exposure and toxicity. Even a very highly toxic chemical will not pose a risk if there is no exposure, or very minimal exposure relative to the toxicity. OPP uses a variety of chemical fate and transport data to develop “estimated environmental concentrations” (EECs) from a suite of established models. The development of aquatic EECs is a tiered process.

The first tier screening model for EECs is with the GENEEC program, developed within OPP, which uses a generic site (in Yazoo, MS) to stand for any site in the U. S. The site choice was intended to yield a maximum exposure, or “worst-case,” scenario applicable nationwide, particularly with respect to runoff. The model is based on a 10 hectare watershed that surrounds a one hectare pond, two meters deep. It is assumed that all of the 10 hectare area is treated with the pesticide and that any runoff would drain into the pond. The model also incorporates spray drift, the amount of which is dependent primarily upon the droplet size of the spray. OPP assumes that if this model indicates no concerns when compared with the appropriate toxicity data, then further analysis is not necessary as there would be no effect on the species.

It should be noted that prior to the development of the GENEEC model in 1995, a much more crude approach was used to determining EECs. Older reviews and Reregistration Eligibility Decisions (REDs) may use this approach, but it was excessively conservative and does not provide a sound basis for modern risk assessments. For the purposes of endangered species consultations, we will attempt to revise this old approach with the GENEEC model, where the old screening level raised risk concerns.

When there is a concern with the comparison of toxicity with the EECs identified in GENEEC model, a more sophisticated PRZM-EXAMS model is run to refine the EECs if a suitable scenario has been developed and validated. The PRZM-EXAMS model was developed with widespread collaboration and review by chemical fate and transport experts, soil scientists, and agronomists throughout academia, government, and industry, where it is in common use. As with the GENEEC model, the basic model remains as a 10 hectare field surrounding and draining into a 1 hectare pond. Crop scenarios have been developed by OPP for specific sites, and the model uses site-specific data on soils, climate (especially precipitation), and the crop or site. Typically, site-scenarios are developed to provide for a worst-case analysis for a particular crop in a particular geographic region. The development of site scenarios is very time consuming; scenarios have not yet been developed for a number of crops and locations. OPP attempts to match the crop(s) under consideration with the most appropriate scenario. For some of the older OPP analyses, a very limited number of scenarios were available. As more scenarios become available and are geographically appropriate to selected T&E species, older models used in previous analyses may be updated.

One area of significant weakness in modeling EECs relates to residential uses, especially by homeowners, but also to an extent by commercial applicators. There are no usage data in OPP that relate to pesticide use by homeowners on a geographic scale that would be appropriate for an assessment of risks to listed species. For example, we may know the maximum application rate for a lawn pesticide, but we do not know the size of the lawns, the proportion of the area in lawns, or the percentage of lawns that may be treated in a given geographic area. There is limited information on soil types, slopes, watering practices, and other aspects that relate to transport and fate of pesticides. We do know that some homeowners will attempt to control pests with chemicals and that others will not control pests at all or will use non-chemical methods. We would expect that in some areas, few homeowners will use pesticides, but in other areas, a high percentage could. As a result, OPP has insufficient information to develop a scenario or address the extent of pesticide use in a residential area.

It is, however, quite necessary to address the potential that home and garden pesticides may affect T&E species, even in the absence of reliable data. Therefore, I have developed a hypothetical scenario, by adapting an existing scenario, to address pesticide use on home lawns where it is most likely that residential pesticides will be used outdoors. It is exceedingly important to note that there is no quantitative, scientifically valid support for this modified scenario; rather it is based on my best professional judgement. I do note that the original scenario, based on golf course use, does have a sound technical basis, and the home lawn scenario is effectively the same as the golf course scenario. Three approaches will be used.

First, the treatment of fairways, greens, and tees will represent situations where a high proportion of homeowners may use a pesticide. Second, I will use a 10% treatment to represent situations where only some homeowners may use a pesticide. Even if OPP cannot reliably determine the percentage of homeowners using a pesticide in a given area, this will provide two estimates. Third, where the risks from lawn use could exceed our criteria by only a modest amount, I can back-calculate the percentage of land that would need to be treated to exceed our criteria. If a smaller percentage is treated, this would then be below our criteria of concern. The percentage here would be not just of lawns, but of all of the treatable area under consideration; but in urban and highly populated suburban areas, it would be similar to a percentage of lawns. Should reliable data or other information become available, the approach will be altered appropriately.

It is also important to note that pesticides used in urban areas can be expected to transport considerable distances if they should run off on to concrete or asphalt, such as with streets (e.g., TDK Environmental, 2001). This makes any quantitative analysis very difficult to address aquatic exposure from home use. It also indicates that a no-use or no-spray buffer approach for protection, which we consider quite viable for agricultural areas, may not be particularly useful for urban areas.

Finally, the applicability of the overall EEC scenario, i.e., the 10 hectare watershed draining into a one hectare farm pond, may not be appropriate for a number of T&E species living in rivers or lakes. This scenario is intended to provide a “worst-case” assessment of EECs, but very many T&E fish do not live in ponds, and very many T&E fish do not have all of the habitat surrounding their environment treated with a pesticide. OPP does believe that the EECs from the farm pond model do represent first order streams, such as those in headwaters areas (Effland, et al. 1999). In many agricultural areas, those first order streams may be upstream from pesticide use, but in other areas, or for some non-agricultural uses such as forestry, the first order streams may receive pesticide runoff and drift. However, larger streams and lakes will very likely have lower, often considerably lower, concentrations of pesticides due to more dilution by the receiving waters. In addition, where persistence is a factor, streams will tend to carry pesticides away from where they enter into the streams, and the models do not allow for this. The variables in size of streams, rivers, and lakes, along with flow rates in the llentic waters and seasonal variation, are large enough to preclude the development of applicable models to represent the diversity of T&E species’ habitats. We can simply qualitatively note that the farm pond model is expected to overestimate EECs in larger bodies of water.

Indirect Effects - We also attempt to protect listed species from indirect effects of pesticides. We note that there is often not a clear distinction between indirect effects on a listed species and adverse modification of critical habitat (discussed below). By considering indirect effects first, we can provide appropriate protection to listed species even where critical habitat has not been designated. In the case of fish, the indirect concerns are routinely assessed for food and cover.

The primary indirect effect of concern would be for the food source for listed fish. These are best represented by potential effects on aquatic invertebrates, although aquatic plants or plankton may be relevant food sources for some fish species. However, it is not necessary to protect individual organisms that serve as food for listed fish. Thus, our goal is to ensure that pesticides will not impair populations of these aquatic arthropods. In some cases, listed fish may feed on other fish. Because our criteria for protecting the listed fish species is based upon the most sensitive species of fish tested, then by protecting the listed fish species, we are also protecting the species used as prey.

In general, but with some exceptions, pesticides applied in terrestrial environments will not affect the plant material in the water that provides aquatic cover for listed fish. Application rates for herbicides are intended to be efficacious, but are not intended to be excessive. Because only a portion of the effective application rate of an herbicide applied to land will reach water through runoff or drift, the amount is very likely to be below effect levels for aquatic plants. Some of the applied herbicides will degrade through photolysis, hydrolysis, or other processes. In addition, terrestrial herbicide applications are efficacious in part, due to the fact that the product will tend to stay in contact with the foliage or the roots and/or germinating plant parts, when soil applied. With aquatic exposures resulting from terrestrial applications, the pesticide is not placed in immediate contact with the aquatic plant, but rather reaches the plant indirectly after entering the water and being diluted. Aquatic exposure is likely to be transient in flowing waters. However, because of the exceptions where terrestrially applied herbicides could have effects on aquatic plants, OPP does evaluate the sensitivity of aquatic macrophytes to these herbicides to determine if populations of aquatic macrophytes that would serve as cover for T&E fish would be affected.

For most pesticides applied to terrestrial environment, the effects in water, even lentic water, will be relatively transient. Therefore, it is only with very persistent pesticides that any effects would be expected to last into the year following their application. As a result, and excepting those very persistent pesticides, we would not expect that pesticidal modification of the food and cover aspects of critical habitat would be adverse beyond the year of application. Therefore, if a listed salmon or steelhead is not present during the year of application, there would be no concern. If the listed fish is present during the year of application, the effects on food and cover are considered as indirect effects on the fish, rather than as adverse modification of critical habitat.

Designated Critical Habitat - OPP is also required to consult if a pesticide may adversely modify designated critical habitat. In addition to the indirect effects on the fish, we consider that the use of pesticides on land could have such an effect on the critical habitat of aquatic species in a few circumstances. For example, use of herbicides in riparian areas could affect riparian vegetation, especially woody riparian vegetation, which possibly could be an indirect effect on a listed fish. However, there are very few pesticides that are registered for use on riparian vegetation, and the specific uses that may be of concern have to be analyzed on a pesticide by pesticide basis. In considering the general effects that could occur and that could be a problem for listed salmonids, the primary concern would be for the destruction of vegetation near the stream, particularly vegetation that provides cover or temperature control, or that contributes woody debris to the aquatic environment. Destruction of low growing herbaceous material would be a concern if that destruction resulted in excessive sediment loads getting into the stream, but such increased sediment loads are insignificant from cultivated fields relative to those resulting from the initial cultivation itself. Increased sediment loads from destruction of vegetation could be a concern in uncultivated areas. Any increased pesticide load as a result of destruction of terrestrial herbaceous vegetation would be considered a direct effect and would be addressed through the modeling of estimated environmental concentrations. Such modeling can and does take into account the presence and nature of riparian vegetation on pesticide transport to a body of water.

Risk Assessment Processes - All of our risk assessment procedures, toxicity test methods, and EEC models have been peer-reviewed by OPP's Science Advisory Panel. The data from toxicity tests and environmental fate and transport studies undergo a stringent review and validation process in accordance with "Standard Evaluation Procedures" published for each type of test. In addition, all test data on toxicity or environmental fate and transport are conducted in accordance with Good Laboratory Practice (GLP) regulations (40 CFR Part 160) at least since the GLPs were promulgated in 1989.

The risk assessment process is described in "Hazard Evaluation Division - Standard Evaluation Procedure - Ecological Risk Assessment" by Urban and Cook (1986) (termed Ecological Risk Assessment SEP below), which has been separately provided to National Marine Fisheries Service staff. Although certain aspects and procedures have been updated throughout the years, the basic process and criteria still apply. In a very brief summary: the toxicity information for various taxonomic groups of species is quantitatively compared with the potential exposure information from the different uses and application rates and methods. A risk quotient of toxicity divided by exposure is developed and compared with criteria of concern. The criteria of concern presented by Urban and Cook (1986) are presented in Table 2.

Table 2.   Risk quotient criteria for direct and indirect effects on T&E fish

Test data	Risk quotient	Presumption
Acute LC_{50}	>0.5	Potentially high acute risk
Acute LC_{50}	>0.1	Risk that may be mitigated through restricted use classification
Acute LC_{50}	>0.05	Endangered species may be affected acutely, including sublethal effects
Chronic NOEC	>1	Chronic risk; endangered species may be affected chronically, including reproduction and effects on progeny
Acute invertebrate LC_{50}[a]	>0.5	May be indirect effects on T&E fish through food supply reduction

Aquatic plant acute EC_{50}[a]	>1[b]	May be indirect effects on aquatic vegetative cover for T&E fish

a. Indirect effects criteria for T&E species are not in Urban and Cook (1986); they were developed subsequently.
b. This criterion has been changed from our earlier requests. The basis is to bring the endangered species criterion for indirect effects on aquatic plant populations in line with EFED's concern levels for these populations.

The Ecological Risk Assessment SEP (pages 2-6) discusses the quantitative estimates of how the acute toxicity data, in combination with the slope of the dose-response curve, can be used to predict the percentage mortality that would occur at the various risk quotients. The discussion indicates that using a "safety factor" of 10, as applies for restricted use classification, one individual in 30,000,000 exposed to the concentration would be likely to die. Using a "safety factor" of 20, as applies to aquatic T&E species, would exponentially increase the margin of safety. It has been calculated by one pesticide registrant (without sufficient information for OPP to validate that number), that the probability of mortality occurring when the LC50 is 1/20th of the EEC is 2.39×10^{-9}, or less than one individual in ten billion. It should be noted that the discussion (originally part of the 1975 regulations for FIFRA) is based upon slopes of primarily organochlorine pesticides, stated to be 4.5 probits per log cycle at that time. As organochlorine pesticides were phased out, OPP undertook an analysis of more current pesticides based on data reported by Johnson and Finley (1980), and determined that the "typical" slope for aquatic toxicity tests for the "more current" pesticides was 9.95. Because the slopes are based upon logarithmically transformed data, the probability of mortality for a pesticide with a 9.95 slope is again exponentially less than for the originally analyzed slope of 4.5.

The above discussion focuses on mortality from acute toxicity. OPP is concerned about other direct effects as well. For chronic and reproductive effects, our criteria ensures that the EEC is below the no-observed-effect-level, where the "effects" include any observable sublethal effects. Because our EEC values are based upon "worst-case" chemical fate and transport data and a small farm pond scenario, it is rare that a non-target organism would be exposed to such concentrations over a period of time, especially for fish that live in lakes or in streams (best professional judgement). Thus, there is no additional safety factor used for the no-observed-effect-concentration, in contrast to the acute data where a safety factor is warranted because the endpoints are a median probability rather than no effect.

Sublethal Effects - With respect to sublethal effects, Tucker and Leitzke (1979) did an extensive review of existing ecotoxicological data on pesticides. Among their findings was that sublethal effects as reported in the literature did not occur at concentrations below one-fourth to one-sixth of the lethal concentrations, when taking into account the same percentages or numbers affected, test system, duration, species, and other factors. This was termed the "6x hypothesis". Their review included cholinesterase inhibition, but was largely oriented towards externally observable parameters such as growth, food consumption, behavioral signs of intoxication, avoidance and repellency, and similar parameters. Even reproductive parameters fit into the hypothesis when the duration of the test was considered. This hypothesis supported the use of lethality tests for use in assessing acute ecotoxicological risk, and the lethality tests are well enough established and understood to provide strong statistical confidence, which can not always be achieved with sublethal effects. By providing an appropriate safety factor, the concentrations found in lethality tests can therefore generally be used to protect from sublethal effects. As discussed earlier, the entire focus of the early-life-stage and life-cycle chronic tests is on sublethal effects.

In recent years, Moore and Waring (1996) challenged Atlantic salmon with diazinon and observed effects on olfaction as relates to reproductive physiology and behavior. Their work indicated that diazinon could have sublethal effects of concern for salmon reproduction. However, the nature of their test system, direct exposure of olfactory rosettes, could not be quantitatively related to exposures in the natural environment. Subsequently, Scholz et al. (2000) conducted a non-reproductive behavioral study using whole Chinook salmon in a model stream system that mimicked a natural exposure that is far more relevant to ecological risk assessment than the system used by Moore and Waring (1996). The Scholz et al. (2000) data indicate potential effects of diazinon on Chinook salmon behavior at very low levels, with statistically significant effects at nominal diazinon exposures of 1 ppb, with apparent, but non-significant effects at 0.1 ppb.

It would appear that the Scholz et al (2000) work contradicts the 6x hypothesis for acute effects. The research design, especially the nature and duration of exposure, of the test system used by Scholz et al (2000), along with a lack of dose-response, precludes comparisons with lethal levels in accordance with the 6x hypothesis as used by Tucker and Leitzke (1979). Nevertheless, it is known that olfaction is an exquisitely sensitive sense. And this sense may be particularly well developed in salmon, as would be consistent with its use by salmon in homing (Hasler and Scholz, 1983). So the contradiction of the 6x hypothesis is not surprising. As a result of these findings, the 6x hypothesis needs to be re-evaluated with respect to olfaction. At the same time, because of the sensitivity of olfaction and because the 6x hypothesis has generally stood the test of time otherwise, it would be premature to abandon the hypothesis for other acute sublethal effects until there are additional data.

2. Description of Triclopyr:

A. Chemical History: Triclopyr BEE was first registered in 1980 for use in non-crop areas and forests for control of broadleaf weeds and woody plants. In 1984 it was registered for use on turf sites. In 1985 BEE was registered for use on rangeland and permanent grass pastures.

B: Chemical Description:

- Common Name: Triclopyr butoxyethyl ester (BEE)
- Chemical Name: Triclopyr core:
  Triclopy[((3,5,6-triclror-2-pyridinyl)oxy)acetic acid]

- ❑ Chemical Family: Pyridinyloxyacetic acids
- ❑ Case Number: 2710
- ❑ CAS Registry Number: 64700-56-7
- ❑ OPP Chemical Code: 116004
- ❑ Molecular Weight: 356.6
- ❑ Empirical Formula: $C_{19}H_{16}CL_2NO_4$
- ❑ Trade and Other Names: Garlon®, Pathfinder®
- ❑ Basic Manufacturer: DowElanco

C. Chemical Use: The following is based on the currently registered uses of triclopyr:

- ❑ Type of Agent: Herbicide
- ❑ Classification: Restricted and Non-Retricted use herbicide (various formulations)

Tryclopyr is a fluffy colorless solid with a melting point of148-150° C. BEE is an oil-soluble liquid which is soluble in acetonitrile, methanol, and n-hexane at >70% by weight.

- ❑ Summary of Sites:
  - ‣ Terrestrial Food/Feed Crops: Pasture and rangeland
  - ‣ Terrestrial Non-Food and Feed Crop: Rights-of-way, turf
  - ‣ Forestry: Conifer Forests

    Public Health: None
  - ‣ Target Pests: (Pacific Northwest and California)
    Vine maple, bigleaf maple, alder, willow, madrone, chinquapin, *Ceanothus sp.*, and undesirable hardwood evergreens..
- ❑ Formulation Types Registered: Formulation intermediate, emulsifiable concentrate, ready-to-use sprayers

<u>Technical Grade/Manufacturing-Use Product (MUP)</u> Triclopyr Butoxyethyl Ester, DowElanco
<u>End-use Product</u> Pathfinder II®, Triclopyr 4 Ester R&P®, Garlon 3A®, Garlon 4®

- Methods of Application:

  - <u>Equipment</u>: airplane, helicopter, ground spreader, backpack sprayers

  - <u>Method and Rate</u>:
    Broadcast
        Ground (GB)
        Aerial (AA)
    High Volume Foliar (HVF)
    Low Volume Foliar (LVF)
    Individual Plant Treatment (IPT)

- Rates of Application - Forestry, reforestation, tree farms, tree plantations

  - BROADCAST, spring, backpack: 3 lb a.i./A

  - DIRECT SPRAY, foliar, low volume, ground: 8.6 lb a.i./A

  Forest tree management, forest pest management

  - DIRECT SPRAY, foliar, low volume, ground: 8.6 lb a.i./A

  Forest tree, unspecified, foliar, low volume, ground: 8.6 lb a.i./A

The above listed rates are the maximum rates identified in the RED. The manufacturer labels, specific to the Pacific Northwest and California identify the maximum rate for forest application as 2 lb a.i./A.

The EPA estimated usage of Triclopy, both acid and ester form, is shown below:

Table 3: Average Annual Triclopy Usage by Site 1987-1995

Site	Acres Grown (X1000)	Acres Treated (X1000)	Percentage Treated	Pounds a.i. Applied (X1000)
Pasture	120,387	327	0.5	292
Woodland	**62,825**	**126**	**0.2**	**100**

Rights of Way	3,200	75	2.3	85
Rice	2,021	165	5.6	77
Railroad	1,080	90	8.5	45
Commercial/ Residential	32,700	75	0.2	40
Other	24,815	88	0.3	34

D. Environmental Fate: Tryclopyr BEE will persist in the environment as the ester for only a limited time. BEE hydrolyzed rapidly to triclopyr acid in natural waters (pH 6.7) with a half-life of 0.5 days. In silty loam, silty clay loam and sandy loam, BEE degrades to tryclopy acid with a half life of about three hours. In all 3 soils, less than 3.2% of the applied BEE remained after 48 hours.

In aerobic soil and water the primary degradation pathway is hydrolysis to triclopyr acid and 2-butoxyethanol, with hydrolysis occurring more rapidly at higher pH
2-butoxyethanol is then rapidly degraded by microbial processes to 2butoxyacetic acid. (half-life of 0.6-3.4 days in water) with the final degradate CO_2. Triclopyr acid and triclopyr are essentially non-toxic to fish and invertebrates. Although triclopyr acid is more persistent than BEE, it does not accumulate in organisms. Triclopyr is moderately persistent, increasing in anerobic conditions, however because it is not expected to reach high concentrations and is non-toxic the Agency concludes that it is not a concern. Triclopyr is not currently regulated under the Safe Drinking Water Act (SDWA), therefore a Maximum Contaminant Level (MCL) has not been set.

Photodegradation of BEE (^{14}C labeled pyridine ring) applied at 1.0 ppm degraded with a half life of 6.6 days under sterile conditions, at buffered pH 6.6, irradiated outdoors in California for 30 days. The major degradate was $^{14}CO_2$, which totaled 29.4%. The other, non-volatile degradates included (5/6)-chloro-3-hydroxy-s-pyrdinone (16%) and dichloropyridinonedynyloxyacetic acid, 2-hydoxyethyl ester (6%). At least 15 other degradates were present, comprising 10% of the applied radioactivity, but were not identified.

Under anerobic conditions (flooded sandy loam soil under nitrogen) BEE degraded quantitatively to triclopyr acid in approximately 5 hours. The acid then was persistent with a half life of approximately 1300 days.

BEE was tested under forestry conditions and was aerially applied at a nominal rate of 3.84 kg ae/ha (Garlon 4, 480 g ae/L) to a forested site (trembling aspen and balsam poplar) in Ontario Canada. Residues were recovered from water as BEE. This was determined to be from over spray of the streams and transport between sampling locations. In stream water, BEE hydrolyzed to tryclopyr acid in 4-6 hours. The maximum observed concentration was 0.35 ppm.

BEE was applied by helicopter to clear-cut timberland in southwest Washington state at a rate of 6 lb a.i/A (Garlon 4; 4 lb a.i./gallon). Only total triclopyr and 3,5,6-trichloro-2-pyridinol (TCP) were detected in streams and stream sediments, although BEE was not specifically analyzed.

The table below illustrates the persistence and mobility of BEE.

Table 4. Mobility and Persistence of Tryclopyr BEE Relative to Restricted Use Criteria

Factor	Characteristic	Triclopyr BEE
Persistence	Field dissipation half-life	0.2 wks (1.1d)
Persistence	Lab derived aerobic soil half-life	NA
Persistence	Hydrolysis half-life	18.172 /5,000%/30 d
Persistence	Photolysis half-life	NA
Mobility	Soil adsorption K_d	NA
Mobility	Soil adsorption K_{OC}	NA
Mobility	Depth of leaching in field dissipation study	45 cm

E. Incidents: A total of 65 incidents have been recorded in the Agency's database. Most are related to human exposure or damage to desirable plants. One pet injury following lawn treatment was noted and there was one fish kill in 1993. This incident involved treatment of a railroad right-of-way with direct runoff into surface water.

F. Estimated and actual concentrations of triclopyr in water: An analysis of toxicity, whether acute or chronic, lethal or sublethal, must be combined with an analysis of how much chemical will be in the water, to determine risks to fish. Risk is a combination of exposure and toxicity. Even a very highly toxic chemical will not pose a risk if there is no exposure, or very minimal exposure relative to the toxicity. OPP uses a variety of chemical fate and transport data to develop "estimated environmental concentrations" (EECs) from a suite of established models.

The Tier II screening models PRZM and EXAMS with the Index Reservoir (IR) and Percent Crop Area adjustments (IR-PCA PRZM/EXAMS) were used to determine estimated surface water concentration of triclopyr. The index reservoir represents a potentially vulnerable drinking water source based on the geometry of an actual reservoir and its watershed (located in Illinois). The PCA is a generic watershed based adjustment factor which represents the portion of a watershed planted to a crop and will be applied to pesticide concentration estimates for surface water exposure.

In an article by Thompson, *et al*, (1995) the environmental fate and ecological fate of BEE were studied in a first order forest stream in Ontario Canada. Maximum concentrations of BEE in stream water samples were 0.848 and 0.949 µg/ml at 2 sites at 10 and 20 minutes after direct injection of BEE into the stream. BEE dissipated rapidly, with stream concentrations decreasing to below 0.1 µg/ml within 50-70 minutes. The authors concluded that in flowing water systems there is rapid dissipation of Triclopy BEE.

In the Agency calculated Estimated Environmental Concentration model (EEC), TEA salt was used, since it dissolves rapidly in water to triclopyr acid and TEA. It was assumed that only the acid would be present in runoff from treated areas. The following parameters, in addition to chemical data, were used to calculated the EECs:

Soil Organic Carbon Partitioning Coefficient (K_{OC}): 204
Aerobic soil metabolism half-life: 18 days
Aerobic aquatic metabolism half-life:142 days
Photolysis half-life (pH 7): 0.6 days
Water Solubility: 440 ppm

Table 5: Estimated Environmental Concentrations (EEC) for Triclopyr TEA

Ground Application

Rate (lb a.i./A)	Peak EEC (ppb)	Day 27 EEC (ppb)	Day 56 EEC (ppb)
1.0	30	25	10
3.128	95	80	61
9.0	270	227	233
12.12	364	305	223
Aerial Application			
6.0	186	156	119

The environmental fate data set is incomplete for Triclopyr BEE ester; no data were available for either the aerobic soil metabolism or aerobic aquatic half-life. However it was possible to generate GENEEC values by making the worst-case assumption that triclopyr BEE was stable to aerobic soil metabolism. The Koc was a reported estimate for BEE (Meylan and Howard, 1992). The following values were used as input for the GENECC Model for triclopyr BEE:

Soil Organic Carbon Partitioning Coefficient (K_{OC}): 560
Aerobic soil metabolism half-life: Stable (GENECC input = 0)
Aerobic aquatic metabolism half-life: No data (GENECC input = 0)
Abiotic hydrolysis half-life (at pH7): 8.7 days
Photolysis half-life: 6.6 days

Water solubility: 6.84 ppm

Table 6: GENECC Aquatic Estimated Environmental Concentration for Triclopyr BEE

Ground Spray

Rate (lbs a.i./A)	Peak EEC (ppb)
1.0	19
3.0	57
8.0	152
12.0	228
Aerial	
1.5	30
8.0	160

G. Ecological Effects Toxicity Assessment:

i. Freshwater Fish: The minimum data required to establish the toxicity of triclopyr technical (for formulation) to freshwater fish is from two species. The preferred species are rainbow trout (coldwater species) and bluegill sunfish (warm water species). Results of these tests are shown in Table 5.

Table 6: Freshwater Fish, Acute Tonicity of Triclopyr BEE

Species	% a.i.	LC_{50} (ppm)	Toxicity Class
Oncorhynchus mykiss (Rainbow trout)	98.98	0.65	highly toxic
Oncorhynchus mykiss (Rainbow trout)	formulated	1.29	moderately toxic
Oncorhynchus mykiss (Rainbow trout)	62.9	0.77-2.7(24hrs)	Moderate to highly toxic
Lepomis macrochirus (Bluegill sunfish)	formulated	1.46	moderately toxic
Lepomis macrochirus (Bluegill sunfish)	98.98	0.36	highly toxic
Lepomis macrochirus (Bluegill sunfish)	62.9	1.3	moderately toxic

Oncorhycus kissutch (Coho salmon)	99	Yolk-sac fry: 0.45-0.47 Juvenile fry 1.4	Yolk-sac fry: highly toxic Juvenile fry: moderately toxic
Plimephales promelas (Fathead minnow)	96.4	2.4 (24 hours)	moderately toxic
Plimephales promelas (Fathead minnow)	96	2.31 (24 hours)	moderately toxic

The results of these studies indicate that triclopyr BEE is moderately to highly toxic to freshwater fish.

ii. Freshwater Fish, Chronic: A freshwater fish early life-cycle test was required for triclopyr because the exposure may be continuous, recurrent, or multiple, due to multiple applications. The results of this testing are shown in Table 7

Table 7: Chronic Tonicity of Triclopyr TEA, Early Life Cycle

Species	% a.i.	MATC(ppm)	Factors Affected	NOEC/LOAC (ppm)
Oncorhynchus mykiss (Rainbow trout)	44.9	130	length	NOAC >104 LOAC <162

The results of these studies indicate that triclopyr may affect the length at concentrations above 104 ppm.

iii. Freshwater Invertebrates, Acute: The preferred species for testing triclopyr toxicity in freshwater invertebrates is the waterflea. Results of acute toxicity tests are shown in Table 8:

Table 8: Acute Toxicity of Triclopyr BEE in Freshwater Invertebrates.

Species	% a.i.	LC_{50}/EC_{50} (ppm)	Toxicity Class
Daphnia magna (waterflea)	96.4	1.7 (nominal conc.)	moderately toxic
Daphnia magna (waterflea)	98.6	12.0	slightly toxic

These studies indicate that triclopyr BEE is slightly to moderately toxic to freshwater invertebrates

iv. Freshwater Invertebrates, Chronic Toxicity: A freshwater invertebrate, early life - cycle test is required for triclopyr due to acute toxicity and potential for transport to water. Results of this testing are shown in Table 9.

Table 9: Chronic Toxicity of Triclopyr to Freshwater Invertebrates

Species	% a.i.	NOEL/ LOEC (ppm)	Factors Affected	MATC (ppm)
Daphnia magna (waterflea)	98.4	80.7 149.0	Total young and brood size	100

These studies indicate that invertebrate reproduction may be impaired at concentrations above 80.7 ppm.

v. Estuarine and Marine Fish, Acute Toxicity: Toxicity testing of triclopyr BEE in marine/estuarine fish was not available at the time of this review.

vi. Estuarine/Marine Fish, Chronic Toxicity: Estuarine/marine fish chronic toxicity, early life-cycle testing of triclopyr BEE data were not available at the time of this review.

vii. Estuarine and Marine Invertebrate Acute Toxicity: Testing was performed to determine the acute toxicity of triclopyr on marine/estuarine invertebrates. The preferred species are mysid shrimp and eastern oyster. Results are shown in Table 11.

Table 10: Acute Toxicity of Triclopyr BEE to Marine/Estuarine Invertebrates

Species	% a.i.	LC_{50}/EC_{50} (ppm)	Toxicity Class
Crassostrea virginica (oyster, shell deposition)	96.1	Not given	highly toxic
Crassostrea virginica (oyster, shell deposition)	62.9	0.32	highly toxic
Palaemonetes pugio (grass shrimp)	96.1	2.47	moderately toxic
Palaemonetes pugio (grass shrimp)	62.4	1.7	moderately toxic
Menidia beryllina (Tidewater silverside)	96.1	0.76	highly toxic
Menidia beryllina (Tidewater silverside)	62.9	0.76	highly toxic

These studies indicate that triclopyr BEE is moderately to highly toxic to marine/estuarine invertebrates.

viii. Estuarine/Marine Invertebrates, Chronic Toxicity: Testing data for chronic toxicity of triclopyr BEE were not requested because it is not expected to be continuous or recurrent in that ecosystem.

The general characterization of triclopyr toxicity for fresh water and marine/estuarine organisms is that it ranges from moderately toxic to highly toxic. A single instance of slightly toxic findings in the water flea does not appear consistent with all other results.

H. Risk Quotients for Subject Species:

Based on toxicity and EEC data, risk quotients were calculated relevant to the T&E species of interest in California and Pacific Northwest ESUs. The results of these calculations are presented in Table 11. The EECs presented are those considered significant to forestry and are estimated by direct application to 6" of lentic water.

Table 11: Risk Quotient Determinations for Aquatic Organisms - Forestry

Site/Method	Application Rate (lbs a.i./A)	Peak EEC (ppm)	Acute RQ	56 Day EEC (ppm)	Chronic RQ
Forest/Aerial	4 lb	2.036	<0.05	0.233	<1

I. Discussion and Characterization of Risk Assessment.

Triclopyr is categorized as being slightly to highly toxic across the spectrum of species tested. BEE is, however, present in the natural aquatic environment for a relatively short period (<4-6 hours) and is rapidly dissipated. The acid form, which is the primary degradate of triclopyr BEE has previously been reviewed by the Agency (memorandum, L. Turner, 2004) and determined to be essentially non-toxic to the species of concern for this report. This suggests that any effects of concern would be of short duration and limited impact. The most recent RED for triclopyr BEE concludes that use of this chemical in forestry poses an acute risk through the direct application to surface waters within the treatment area. These concerns are reflected in the EEC and RQ determinations. With respect to this review, however, it is noted that small, intermittent streams in the ESU's of concern are not compatible with the known behavior patterns of the species of interest.

J. Existing Protections: Currently the expected precautions regarding spray drift and personal safety measures are components of the label language for triclopyr. In addition, specific measures are included regarding application rates based on geographic location. Direct aquatic application is prohibited. Other restrictions are provided for homeowner use.

K. Proposed Protections. For forestry use the maximum application rate is reduced to 8 lbs a.i./A. Spray Drift Task Force guidelines are included. "Double notification" requirements (notification in writing and orally) to workers have been added..

3. Description of Pacific salmon and steelhead Evolutionarily Significant Units relative to Specific Usage by County:

The data for counties within California were take directly from the California Department of Pesticide Regulation agricultural pesticide use tables within the 2002 reporting period. The values indicated are each reported treatment areas and pounds of active ingredient used for each county. Because Oregon does not provide such data, the data presented for Oregon are calculated values. The area treated represents an estimate of total forested area based on US Geological Service mapping by plainametric approximation of county size and the Oregon Department of Forestry that 75% of that area is forested. This was corrected to the EPA estimated use of 0.3%. To determine pounds applied, the maximum label application for forest application, 2 lbs a.i./A, was used.

A. Steelhead

Steelhead, *Oncorhyncus mykiss*, exhibit one of the most complex suite of life history traits of any salmonid species. Steelhead may exhibit anadromy or freshwater residency. Resident forms are usually referred to as ‘‘rainbow’’ or ‘‘redband’’ trout, while anadromous life forms are termed ‘‘steelhead.’’ The relationship between these two life forms is poorly understood, however, the scientific name was recently changed to represent that both forms are a single species.

Steelhead typically migrate to marine waters after spending 2 years in fresh water. They then reside in marine waters for typically 2 or 3 years prior to returning to their natal stream to spawn as 4- or 5-year-olds. Unlike Pacific salmon, they are capable of spawning more than once before they die. However, it is rare for steelhead to spawn more than twice before dying; most that do so are females. Steelhead adults typically spawn between December and June. Depending on water temperature, steelhead eggs may incubate in redds for 1.5 to 4 months before hatching as alevins. Following yolk sac absorption, alevins emerge as fry and begin actively feeding. Juveniles rear in fresh water from 1 to 4 years, then migrate to the ocean as ‘‘smolts.’’

Biologically, steelhead can be divided into two reproductive ecotypes. "Stream maturing," or "summer steelhead" enter fresh water in a sexually immature condition and require several months to mature and spawn. "Ocean maturing," or "winter steelhead" enter fresh water with well-developed gonads and spawn shortly after river entry. There are also two major genetic groups, applying to both anadromous and non-anadromous forms: a coastal group and an inland group, separated approximately by the Cascade crest in Oregon and Washington. California is thought to have only coastal steelhead while Idaho has only inland steelhead.

Historically, steelhead were distributed throughout the North Pacific Ocean from the Kamchatka Peninsula in Asia to the northern Baja Peninsula, but they are now known only as far south as the Santa Margarita River in San Diego County. Many populations have been extirpated.

1. Central California Coast Steelhead ESU

The Central California coast steelhead ESU was proposed for listing as endangered on August 9, 1996 (61FR41541-41561) and the listing was made final, as threatened, a year later (62FR43937-43954, August 18, 1997). Critical Habitat was proposed February 5, 1999 (64FR5740-5754) and designated on February 16, 2000 (65FR7764-7787). This coastal steelhead ESU occupies California river basins from the Russian River, Sonoma County, to Aptos Creek, Santa Cruz County, (inclusive), and the drainage of San Francisco and San Pablo Bays eastward to the Napa River (inclusive), Napa County. The Sacramento-San Joaquin River Basin of the Central Valley of California is excluded. Steelhead in most tributary streams in San Francisco and San Pablo Bays appear to have been extirpated, whereas most coastal streams sampled in the central California coast region do contain steelhead.

Only winter steelhead are found in this ESU and those to the south. River entry ranges from October in the larger basins, late November in the smaller coastal basins, and continues through June. Steelhead spawning begins in November in the larger basins, December in the smaller coastal basins, and can continue through April with peak spawning generally in February and March. Hydrologic units in this ESU include Russian (upstream barriers - Coyote Dam, Warm Springs Dam), Bodega Bay, Suisun Bay, San Pablo Bay (upstream barriers - Phoenix Dam, San Pablo Dam), Coyote (upstream barriers - Almaden, Anderson, Calero, Guadelupe, Stevens Creek, and Vasona Reservoirs, Searsville Lake), San Francisco Bay (upstream barriers - Calveras Reservoir, Chabot Dam, Crystal Springs Reservoir, Del Valle Reservoir, San Antonio Reservoir), San Francisco Coastal South (upstream barrier - Pilarcitos Dam), and San Lorenzo-Soquel (upstream barrier - Newell Dam).

Counties of occurrence for this ESU are Santa Cruz, San Mateo, San Francisco, Marin, Sonoma, Mendocino, Napa, Alameda, Contra Costa, Solano, and Santa Clara counties. Usage of Triclopyr in the counties where the Central California coast steelhead ESU is presented in Table.

Counties supporting the Central California Coast steelhead ESU

County	Crop(s)	Acres Treated	Pounds Applied
Alameda	Forest Trees		None
Contra Costa	Forest Trees		None
Marin	Forest Trees		None
Mendocino	Forest Trees	250	154

Napa	Forest Trees		None
San Francisco	Forest Trees		None
San Mateo	Forest Trees		None
Santa Clara	Forest Trees		None
Santa Cruz	Forest Trees		None
Solano	Forest Trees		None
Sonoma	Forest Trees	100	39

The use of triclopyr BEE in the Central California Steelhead ESU is minimal. The use of this chemical at the reported rates will have no effect on the endangered species of concern for this review.

B. Chinook salmon

Chinook salmon *(Oncorhynchus tshawytscha)* is the largest salmon species; adults weighing over 120 pounds have been caught in North American waters. Like other Pacific salmon, chinook salmon are anadromous and die after spawning.

Juvenile stream- and ocean-type chinook salmon have adapted to different ecological niches. Ocean-type chinook salmon, commonly found in coastal streams, tend to utilize estuaries and coastal areas more extensively for juvenile rearing. They typically migrate to sea within the first three months of emergence and spend their ocean life in coastal waters. Summer and fall runs predominate for ocean-type chinook. Stream-type chinook are found most commonly in headwater streams and are much more dependent on freshwater stream ecosystems because of their extended residence in these areas. They often have extensive offshore migrations before returning to their natal streams in the spring or summer months. Stream-type smolts are much larger than their younger ocean-type counterparts and are therefore able to move offshore relatively quickly.

Coast-wide, chinook salmon typically remain at sea for 2 to 4 years, with the exception of a small proportion of yearling males (called jack salmon) which mature in freshwater or return after 2 or 3 months in salt water. Ocean-type chinook salmon tend to migrate along the coast, while stream-type chinook salmon are found far from the coast in the central North Pacific. They return to their natal streams with a high degree of fidelity. Seasonal ‘‘runs’’ (i.e., spring, summer, fall, or winter), which may be related to local temperature and water flow regimes, have been identified on the basis of when adult chinook salmon enter freshwater to begin their spawning migration. Egg deposition must occur at a time to ensure that fry emerge during the following spring when the river or estuarine productivity is sufficient for juvenile survival and growth.

Adult female chinook will prepare a spawning bed, called a redds, in a stream area with suitable gravel composition, water depth and velocity. After laying eggs in a Redds, adult chinook will guard the Redds from 4 to 25 days before dying. Chinook salmon eggs will hatch, depending upon water temperatures, between 90 to 150 days after deposition. Juvenile chinook may spend from 3 months to 2 years in freshwater after emergence and before migrating to estuarine areas as smolts, and then into the ocean to feed and mature. Historically, chinook salmon ranged as far south as the Ventura River, California, and their northern extent reaches the Russian Far East.

2. California Coastal Chinook Salmon ESU

The California coastal chinook salmon ESU was proposed as threatened in 1998 (63FR11482-11520, March 9, 1998) and listed on September 16, 1999 (64FR50393-50415). Critical habitat was designated February 16, 2000 (65FR7764-7787) to encompass all river reaches and estuarine areas accessible to listed chinook salmon from Redwood Creek (Humboldt County, California) to the Russian River (Sonoma County, California), inclusive.

The Hydrologic units and upstream barriers are Mad-Redwood, Upper Eel (upstream barrier - Scott Dam), Middle Fort Eel, Lower Eel, South Fork Eel, Mattole, Big-Navarro-Garcia, Gualala-Salmon, Russian (upstream barriers - Coyote Dam; Warm Springs Dam), and Bodega Bay. Counties with agricultural areas where Triclopyr BEE could be used are Humboldt, Trinity, Mendocino, Lake, Sonoma, and Marin. A small portion of Glenn County is also included in the Critical Habitat, but triclopyr BEE would not be used in the protected, forested, upper elevation areas.

California counties supporting the California coastal chinook salmon ESU

County	Crop(s)	Acres Treated	Pounds. Applied
Humboldt	Forest Trees	8027	16982
Lake	Forest Trees	547	251
Marin	Forest Trees		None
Mendocino	Forest Trees	250	154
Sonoma	Forest Trees	100	39
Trinity	Forest Trees	650	1258

In focal areas (Humbolt and Trinity Counties) there is significant use of triclopyr BEE within the California Coastal Chinook salmon ESU. There is a potential for focal, transient impact on the species of concern. The short half-life and rapid dispersion of the chemical significantly mitigate major damage, leading to the conclusion that triclopyr BEE use may affect, but is not likely to adversely affect the species of concern.

C. Coho Salmon

Coho salmon, *Oncorhynchus kisutch*, were historically distributed throughout the North Pacific Ocean from central California to Point Hope, AK, through the Aleutian Islands into Asia. Historically, this species probably inhabited most coastal streams in Washington, Oregon, and central and northern California. Some populations may once have migrated hundreds of miles inland to spawn in tributaries of the upper Columbia River in Washington and the Snake River in Idaho.

Coho salmon generally exhibit a relatively simple, 3 year life cycle. Adults typically begin their freshwater spawning migration in the late summer and fall, spawn by mid-winter, then die. Southern populations are somewhat later and spend much less time in the river prior to spawning than do northern coho. Homing fidelity in coho salmon is generally strong; however their small tributary habitats experience relatively frequent, temporary blockages, and there are a number of examples in which coho salmon have rapidly re-colonized vacant habitat that had only recently become accessible to anadromous fish.

After spawning in late fall and early winter, eggs incubate in redds for 1.5 to 4 months, depending upon the temperature, before hatching as alevins. Following yolk sac absorption, alevins emerge and begin actively feeding as fry. Juveniles rear in fresh water for up to 15 months, then migrate to the ocean as ‘‘smolts’’ in the spring. Coho salmon typically spend two growing seasons in the ocean before returning to their natal stream. They are most frequently recovered from ocean waters in the vicinity of their spawning streams, with a minority being recovered at adjacent coastal areas, decreasing in number with distance from the natal streams. However, those coho released from Puget Sound, Hood Canal, and the Strait of Juan de Fuca are caught at high levels in Puget Sound, an area not entered by coho salmon from other areas.

3. Southern Oregon/Northern California Coast Coho Salmon ESU

The Southern Oregon/Northern California coastal coho salmon ESU was proposed as threatened in 1995 (60FR38011-38030, July 25, 1995) and listed on May 6, 1997 (62FR24588-24609). Critical habitat was proposed later that year (62FR62741-62751, November 25, 1997) and finally designated on May 5, 1999 (64FR24049-24062) to encompass accessible reaches of all rivers (including estuarine areas and tributaries) between the Mattole River in California and the Elk River in Oregon, inclusive.

The Southern Oregon/Northern California Coast coho salmon ESU occurs between Punta Gorda, Humboldt County, California and Cape Blanco, Curry County, Oregon. Major basins with this salmon ESU are the Rogue, Klamath, Trinity, and Eel river basins, while the Elk River, Oregon, and the Smith and Mad Rivers, and Redwood Creek, California are smaller basins within the range. Hydrologic units and the upstream barriers are Mattole, South Fork Eel, Lower Eel, Middle Fork Eel, Upper Eel (upstream barrier - Scott Dam-Lake Pillsbury), Mad-Redwood, Smith, South Fork Trinity, Trinity (upstream barrier - Lewiston Dam-Lewiston Reservoir), Salmon, Lower Klamath, Scott, Shasta (upstream barrier - Dwinnell Dam-Dwinnell Reservoir), Upper Klamath (upstream barrier - Irongate Dam-Irongate Reservoir), Chetco, Illinois (upstream barrier - Selmac Dam-Lake Selmac), Lower Rogue, Applegate (upstream barrier - Applegate Dam-Applegate Reservoir), Middle Rogue (upstream barrier - Emigrant Lake Dam-Emigrant Lake), Upper Rogue (upstream barriers - Agate Lake Dam-Agate Lake; Fish Lake Dam-Fish Lake; Willow Lake Dam-Willow Lake; Lost Creek Dam-Lost Creek Reservoir), and Sixes. Related counties are Humboldt, Mendocino, Trinity, Glenn, Lake, Del Norte, Siskiyou in California and Curry, Jackson, Josephine, and Douglas, in Oregon. However, I have excluded Glenn County, California from this analysis because the salmon habitat in this county is not near the agricultural areas where triclopyr can be used. Klammath county is excluded because it lies beyond an impassable barrier.

Table shows the usage of triclopyr BEE in the California counties supporting the Southern Oregon/Northern California coastal coho salmon ESU. Table shows the cropping information for Oregon counties where the Southern Oregon/Northern California coastal coho salmon ESU occurs.

California Counties where the Southern Oregon/Northern California Coastal Coho Salmon ESU Occurs

County	Crop(s)	Acres Treated	Pounds Applied
Del Norte	Forest Trees	623	961
Humboldt	Forest Trees	8027	16982
Lake	Forest Trees	547	251
Mendocino	Forest Trees	250	154
Siskiyou	Forest Trees	757	1099
Trinity	Forest Trees	650	1252

Oregon counties where there is habitat for the Southern Oregon/Northern California coastal coho salmon ESU.

St	County	Crops and acres planted	Acres Treated	Pounds Applied
OR	Curry	Forest Trees	82	164
OR	Douglas	Forest Trees	4275	26
OR	Jackson	Forest Trees	51	102
OR	Josephine	Forest Trees	84	168

In focal areas (Humbolt, Siskiyou and Trinity Counties) there is significant use of triclopyr BEE within the California Coastal Chinook salmon ESU. There is a potential for focal, transient impact on the species of concern. The short half-life and rapid dispersion of the chemical significantly mitigate major damage, leading to the conclusion that triclopyr BEE use may affect, but is not likely to adversely affect the species of concern.

Summary of Review:

Species	ESU	Finding
Chinook Salmon	California Coastal	May Affect, not likely to adversely affect
Coho Salmon	Southern Oregon/Northern California	May Affect, not likely to adversely affect
Steelhead	Central California Coast	No Effect

Assessment:

Within the areas of concern there are focal zones where triclopyr BEE is used in significant quantities. This is seen in the California Coastal Chinook Salmon ESU and the Southern Oregon, Northern California Coho Salmon ESU. Due to the high acute toxicity of the chemical, the possibility of an isolated event occurring can not be disregarded. This event, based on noted chemical half life and rapid dissipation, would be of a transient nature and should not have a significant affect on the endangered species. In this regard I refer to the long history of use and the presence of only one recorded fish kill, not associated with forestry application.

In the Central California Coastal Steelhead ESU, a heavily populated location, forestry is not a major activity. Use of triclopyr BEE for forest management is therefore very minimal and will have no affect, transient or otherwise, on the endangered steelhead population.

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Attachment 1
Reregistration Eligibility Decision for Triclopyr

Attachment 2
Sample Product Labels

Attachment 3
USGS Usage Map