Dichlobenil
Analysis of Risks
to
Endangered and Threatened Pacific Salmon and Steelhead

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Summary

Dichlobenil is a benzonitrile herbicide registered nationally for control of weeds and grasses in agricultural, residential and industrial areas. These sites include fruit and nut orchards, cranberries, hybrid poplar-cottonwood plantations, rights-of-ways, woody ornamentals, recreational areas, under asphalt and in sewer lines. A Reregistration Eligibility Document (RED) that includes an ecological risk assessment for fish, invertebrates and aquatic plants was issued in October 1998. Dichlobenil is moderately toxic to freshwater fish and estuarine invertebrates, slightly to moderately toxic to freshwater invertebrates and slightly toxic to estuarine fish. It is also toxic to aquatic vegetation. The Estimated Environmental Concentrations (EECs) were modeled with a Tier 1 model, GENEEC, for all application rates from 2 to 20 pounds a.i./a. Acute and chronic risk quotients calculated from these EECs and the available toxicity values indicate no direct risk to endangered fish or aquatic invertebrate populations and low exposure potential to aquatic plants. We conclude that dichlobenil will not present a direct effect on Pacific salmon and steelhead, no indirect effects based on loss of their aquatic invertebrate food supply and no indirect effects based on loss of aquatic plant cover.

Introduction

Problem formulation: The purpose of this analysis is to determine whether the registration of dichlobenil as an herbicide for use on various treatment sites may affect threatened and endangered (T&E or listed) Pacific anadromous salmon and steelhead and their designated critical habitat.

Scope: Although this analysis is specific to listed Pacific anadromous salmon and steelhead and the watersheds in which they occur, it is acknowledged that dichlobenil 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.

Contents

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. 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 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, we 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. we 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. we 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. we 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.

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 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, 1991). 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.

It is, however, quite necessary to address the potential that home and garden pesticides may have to affect T&E species, even in the absence of reliable data. Therefore, we 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 our best professional judgement. we 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, we 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, we 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.

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 lotic 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 fish and aquatic invertebrates

Test data	Risk quotient	Presumption
Acute LC50	>0.5	Potentially high acute risk
Acute LC50	>0.1	Risk that may be mitigated through restricted use classification
Acute LC50	>0.05	Endangered species may be affected acutely, including sublethal effects
Chronic NOEC	>1.0	Chronic risk; endangered species may be affected chronically, including reproduction and effects on progeny
Acute invertebrate LC50	>0.5	May be indirect effects on T&E fish through food supply reduction
Aquatic plant acute EC50	>1.0	May be indirect effects on aquatic vegetative cover for T&E fish

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 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.

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. 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 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 sublethal effects until there are additional data.

2. Description and use of dichlobenil

a. Description of chemical

Dichlobenil is a benzonitrile herbicide, first registered in 1964, that is used to control weeds and grasses in agricultural, residential and industrial areas. It kills targeted plants by inhibiting seed germination and cell division in meristems. Treatment sites include: cranberry bogs; ornamentals; blackberry, raspberry and blueberry fields; pear, apple, filbert and cherry orchards; vineyards; hybrid poplar-cottonwood plantations for wood and pulp production; and noncrop areas such as rights-of-ways, paved areas, sidewalks, recreational areas and fences. Dichlobenil is used in areas where all vegetation is to be killed, such as around established trees, fences, shrubs and other structures. It is not applied to lawns. The recommended timing for application is early spring and late fall, when cooler temperatures are expected. Cool temperatures would minimize vaporization, as dichlobenil is moderately volatile. Dichlobenil is also used to remove tree roots and inhibit their growth in sewers.

While the current label refers to hybrid poplar-cottonwood plantations in general, a revised label submitted by the registrant requested that the use of dichlobenil be limited in eastern Oregon and Washington. The geographic area will be limited to "the eastern Oregon-Washington desert, defined as 15 miles from the Columbia River in the counties of Walla Walla, Franklin and Benton in Washington and Umatilla and Morrow counties in Oregon."

It is formulated as granulars (1 to 10% active ingredient), liquid-ready to use (0.5% a.i.), soluble concentrate (0.5% a.i.) and wettable powders (0.55% a.i.). The nongranular formulations are applied only as pre-pavement treatment and in sewage systems. Some of the products used in sewers also contain the active ingredient metam sodium. The labels allow for the granules to be applied by air or ground equipment followed by soil incorporation.

b. Summary of labeled uses

<u>General directions</u> – Apply in late fall or very early spring. Use the lower rates of application if treatment is followed by irrigation to incorporate the chemical. Shallow incorporation by mechanical means, rainfall or sprinkler irrigation is suggested when applications are made during periods of high temperatures due to the volatility of the chemical.

<u>Fruit and nut orchards and nurseries– One application per year.</u>

Apple, blueberry, cherry, grape, and filbert – 4 to 6 lb a.i./a.

Blackberry and raspberry – 4 lb a.i./a.

<u>Cranberries</u> – 4 lb a.i./a if only one application or two applications 3 to 6 weeks apart of 2 lb a.i./a each.

<u>Woody ornamentals and shelterbelts (established and in nurseries)</u> – one application of 4

to 8 lb a.i./a. Do not use on pines in California.

Hybrid cottonwood and poplar plantations and stoolbeds – One application of 2 to 4 lb a.i./a.

Noncrop areas (roadways, railroads, around structures, fencerows, utility rights of way, recreational areas, under containerized nursery stock – One application of 4 to 8 lb a.i./a. For nutsedge control apply 10 to 20 lb a.i./a and then incorporate immediately to a depth of 4 to 6 inches.

Under asphalt – One application of 10 to 12 lb a.i./a after the final grade is achieved. The treated area should be covered with asphalt as soon as possible.

Under vinyl pool liners of above ground pools – One application of 12 lb a.i./a followed by incorporation. Up to 0.18 lb a.i./a around perimeter of pool.

Sewer systems (public and residential) – amount applied depends on width of sewer line and distance being treated.

Homeowner user –The homeowner formulations are packaged in shaker containers. The amounts that are shaken on the bare ground for each use are given on the label as " x tablespoons (y oz.) per 10 sq. ft. of area", with the number of tablespoons and ounces varying for each site being treated. We converted the tablespoons and ounces of the formulated products to lb a.i./a for uniformity and comparison with the non-homeowner applications, but it is obvious from the label directions that the herbicide is applied in localized sections around the home, and not spread out in large areas of the lawn. Only one application a year is needed. Label directions tell the user to sprinkle the treated areas lightly with water to moisten the soil 1/4 to ½ inch deep.

Woody ornamentals – 4 to 6 lb a.i./a.

Fruit trees, nuts, blueberries, grapes – 4 to 6 lb a.i./a.

Blackberries, raspberries – 4 to 6 lb a.i./a.

Noncrop areas (around buildings, fences and other structures)– 4 to 6 lb a.i./a.

Noncrop areas (driveways, fence lines, gravel parking areas, walkways, around buildings, under decks) –15 to 18 oz. per 100 square feet or 8 - 10 pounds a.i./a. Irrigation should occur within a few day of application. (This is an amended label, approved June 2002, to comply with changes required in the RED. The RED used the then-registered rates of 1-3/8 to 2-1/4 pounds of 2G product per 100 square feet (12 to 20 pounds a.i./a).

Personal communication from the major registrant (M. Lengen and M. Schocken, Crompton Corp., June 2003) indicated that in the Pacific Northwest dichlobenil is only applied by ground equipment to the apple, berry, cherry, pear and cranberry crops, woody ornamentals and cottonwood-poplar plantations. The berry and cranberry fields in this region are relatively small, and only the areas around and between the tree fruit crops are treated. Therefore aerial applications are not practical in this region. (The information given to us by Crompton Corp. is used in other parts of this document without further citation of the authors.)

The registrant also stated that although the label permits use rates of up to 20 lb a.i./a on noncrop areas, the typical application rate is in the range of 4 to 6 lb a.i./a. The registrant attributes this difference to the high cost of dichlobenil products. Because of the expense the products are typically applied as spot treatments to established landscape plants and around fences and buildings. It is not typically applied to large areas and is not used to treat lawns or other turf grass areas. It is generally applied with a hand-held spreader (called a belly-grinder), and the granules are applied about two feet or less above the ground.

The sewer system treatment is unique and the registrant provided information on the method of application in Washington. The 85% a.i. formulation is used in the sewer systems in Washington from September to June on a periodic basis when the wastewater flows are at the highest level. The product is applied as a foam into sewer pipes whereby dichlobenil is suspended in the water and air is added to the mixture, and a foam is created. Normally less than 2 pounds of 85% dichlobenil enters the wastewater system. The precautionary label statements caution against treating large sections of the pipes to avoid adversely affecting the biological sewage breakdown process in wastewater treatment plants. Treatment of the system should be done at intervals of one to two days. When the pesticide enters the systems some of the dichlobenil sticks to the walls of the pipes and root masses, but the balance enters the wastewater system where the dissolved portion (about 20 ppm) is carried to the wastewater plant while a portion settles out in the pipes and is dissolved and broken down over time. The effluent from the treated sewer pipes flows to a wastewater treatment plant where it is treated prior to discharge to waterways. The registrant indicates that a study that was done in Canada which tested for dichlobenil in a treatment plant’s effluent showed residues less than 0.5 ppb. The citation for the study was not provided.

c. Proposed label changes required by the RED

The 1998 RED required several label changes to reduce ecological risks. These include:

- ground water advisory
- reduction of highest application rate to $\leq$ 10 pounds a.i./a
- soil incorporation of 10% granular formulations

Crompton Corp. submitted a new label incorporating these changes to the Agency. The highest application rate will continue to be for the control of nutsedge in noncrop areas, but at a rate of no more than 10 pounds a.i./a (instead of the 20 pounds a.i./a on the current label) followed immediately by incorporation through watering-in to a depth of 4 to 6 inches. Although it was not required in the RED, the changes to restrict the treatment of cottonwood and poplar plantations to eastern Oregon and Washington are also included on the new label. The reduction of the 20 pounds a.i./a rate to no more than 10 pounds a.i./a will be in effect for all dichlobenil products. The label for the homeowner treatment of driveways and walkways has already been changed to comply with the RED requirements.

d. Estimated usage of dichlobenil

We have no recent national data on the amount of dichlobenil applied annually in the U.S. The RED provided national usage data for 1993 to 1995 indicating that approximately 150,000 to 225,000 pounds active ingredient were used to treat about 55,000 to 95,000 acre treatments in the aggregate, divided into 42,000 to 69,000 agricultural aggregate acres and 15,000 to 25,000 non-agricultural aggregate acres. As stated in the RED the agricultural sites represent nearly 58 percent of the total usage (89,000 to 129,000 pounds a.i.), with use on ornamentals comprising 66 percent of this amount used. Most agricultural use of dichlobenil was on cranberries (15, 000 to 20,000 pounds a.i.); the amounts on the other registered fruit and nut crops were <1,000 to 5,000 pounds a.i. for each individual crop. Other important sites were the homeowner uses which comprised 51 to 62 percent of the non-agricultural total, with 40,000 to 50,000 pounds a.i. used on 10,000 to 15,000 total acre treatments. The other non-agricultural uses such as sewers, rights-of-way and fence rows had no more than 1,000, 2,000 and 5,000 total acre treatments, respectively. Although no one major region or state accounted for the majority of the usage, states in the northwest were listed as major areas of use for the homeowner and rights-of way uses.

Some additional data from the 1990s also are available from the U.S. Geological Survey (USGS). The USGS estimated county pesticide use for the conterminous United States by combining (1) state-level information on pesticide use rates available from the National Center for Food and Agricultural Policy from pesticide use information collected by state and federal agencies over a 4-year period (1992–1995), and (2) county-level information on harvested crop acreage from the 1992 Census of Agriculture. The average annual pesticide use, the total amount of pesticide applied (in pounds), and the corresponding area treated (in acres) were compiled for 208 pesticide compounds that are applied to crops in the conterminous United States. Pesticide use was ranked by compound and crop on the basis of the amount of each compound applied to 86 selected crops. Their data indicate that the agricultural crops of highest dichlobenil usage during the mid-1990s were cranberries (~39,000 lb ai) and apples (~11,000 lb ai). USGS also mapped dichlobenil use on selected crops (Figure 1). This map is included here as a quick and easy visual depiction of where dichlobenil may have been used on agricultural crops. However, it should not be used for any quantitative analysis, because it is based on 1992 crop acreage data and was developed from 1990-1995 statewide estimates of use that were then applied to that county acreage without consideration of local practices and usage. Refer to the attached map from http://ca.water.usgs.gov/pnsp/use92/dichlbnl.html.

At the state and county level, more data are available for dichlobenil use in California than in Oregon, Washington, and Idaho. California requires full pesticide-use reporting by most applicators (excluding homeowners), and the California Department of Pesticide Regulation (DPR) provides the information at the county level (www.cdpr.ca.gov/docs/pur/purmain.htm).

We are not aware of any comprehensive sources of annual pesticide-use information for Oregon, Washington, or Idaho. Oregon is attempting to implement full pesticide-use reporting but has not yet done so.

Information for selected crops in Washington and Oregon is available from the USDA/NASS Washington Agricultural Statistics Service in their "Agricultural Chemical Usage" reports (http://jan.mannlib.cornell.edu/reports/nassr/other/pcu-bb/#fruits) but the data are not reported at the county level. The data for 2001 indicate that dichlobenil use is very small compared to use of other herbicides registered for the same fruit and nut crops as dichlobenil. The report included dichlobenil under the list of herbicides used on blueberries and blackberries in Oregon and raspberries in Oregon and Washington, but the amounts were so limited (less than 50 pounds a.i., the smallest amount they recorded) that no usage data was provided. Dichlobenil was not listed in the tables for apples, cherries, grapes and pears in the report.

Personal communication with Ray Willard, Washington State Department of Transportation, also disclosed that his department uses dichlobenil by means of belly grinders and other hand-held application equipment in urban areas with ornamental landscapes and to treat rights-of-way areas. He stated that dichlobenil is not a significant herbicide in terms of amount applied to these rights of ways, but is primarily used by homeowners and park maintenance as spot treatments. A typical application involves treating 10 to 20 acres over 5 to 10 miles of freeway.

The 2000 and 2001 Annual Reports for California indicates that only 2235 pounds a.i. and 2750 pounds a.i. dichlobenil, respectively, were applied throughout the state.. Table 3 presents the uses and amounts of active ingredient applied in both years.

Table 3. Uses of dichlobenil in California in 2000 and 2001 (Source: California DPR Pesticide Use Report)

2000			2001		
Commodity	**lb a.i.**	**Amount treated**[1]	**Commodity**	**lb a.i.**	**Amount treated**[1]
Commodity fumigation	3				
Grape, wine	160	38 acres	Grape, wine	157	76 acres
Industrial site	2	<1acre	Industrial site	3	<1acre
Landscape maintenance	154		Landscape maintenance	71	
Outdoor plants - containers	984	588 acres	Outdoor plants - containers	1,933	246 acres
Raspberry	4	2 acres			
Regulatory pest control	55				
Residential	2	NA[2]			

Rights of way	435		Rights of way	316	
Structural pest control	420		Structural pest control	263	
Uncultivated ag	10	1 acre			
Uncultivated non-ag	3	NA[2]	Uncultivated non-ag	3	<1acre
			Water area	1	108 acres

[1]Amount treated = cumulative areas or units treated over time with the active ingredient.
[2]NA = Not available

Crompton Corp. submitted information on the amounts of dichlobenil sold in the four states of the Pacific Northwest including California by use site for 2001 and 2002. However, these data are proprietary and are protected by FIFRA definitions of Confidential Business Information (CBI) under Section 10 (d)(1)(A), (B) and (C). However, in general terms, relatively little dichlobenil is used on crops as compared to other herbicides registered for the same sites. The majority of dichlobenil is used on noncrop sites in these states, but there is no information on the sizes of noncrop areas treated.

3. General aquatic risk assessment for endangered and threatened salmon and steelhead

a. Aquatic toxicity

The acute toxicity data indicate that technical grade dichlobenil is moderately toxic to freshwater fish and estuarine invertebrates, slightly to moderately toxic to freshwater invertebrates and slightly toxic to estuarine fish. The primary degradate 2,6 dichlorobenzamide (BAM) is practically nontoxic to freshwater fish and invertebrates. Tests of aquatic invertebrates conducted with a 50% formulation indicated it is also slightly to moderately toxic to this group of animals. The data from the RED and the EFED database are presented in Tables 4 through 9, and the data from the AQUIRE database is presented in Table 10.

Table 4. Acute toxicity of dichlobenil to freshwater fish (source: EFED Pesticide Ecotoxicity Database and RED)

Species	Scientific Name	% ai	96-h LC 50 (ppm)	Toxicity Category
Rainbow trout	*Oncorhynchus mykiss*	98.9	6.26	Moderately toxic
Rainbow trout	*Oncorhynchus mykiss*	98.9	4.93	Moderately toxic
Rainbow trout	*Oncorhynchus mykiss*	99.5	7.6	Moderately toxic
Bluegill sunfish	*Lepomis macrochirus*	98.9	6.72	Moderately toxic
Bluegill sunfish	*Lepomis macrochirus*	98.9	8.31	Moderately toxic
Fathead minnow	*Pimephales promelas*	98.9	6.00	Moderately toxic
Fathead minnow	*Pimephales promelas*	98.9	7.12	Moderately toxic

Goldfish	*Carassius auratus*	98.9	7.83	Moderately toxic
Goldfish	*Carassius auratus*	98.9	7.68	Moderately toxic
Green sunfish	*Lepomis cyanellus*	98.9	5.7	Moderately toxic
Green sunfish	*Lepomis cyanellus*	98.9	12.6	Moderately toxic

Table 5. Acute toxicity of dichlobenil to freshwater invertebrates (source: EFED Pesticide Ecotoxicity Database and RED)

Species	Scientific Name	% ai	96-h LC 50 (ppm)	Toxicity Category
Waterflea	*Daphnia magna*	99.0	6.2	Moderately toxic
Waterflea	*Daphnia magna*	tech.	10.0	Moderately toxic
Sowbug	*Asellus brevicaudus*	tech.	34.0	Slightly toxic
Scud	*Gammarus fasciatus*	tech.	18.0	Slightly toxic
Grass shrimp	*Palaemonetes kadiakensis*	tech.	9.0	Moderately toxic
Seed shrimp	*Cypridopsis vidua*	tech.	7.8	Moderately toxic
Waterflea	*Simocephalus serrulatus*	50.0	5.8	Moderately toxic
Waterflea	*Daphnia pulex*	50.0	3.7	Moderately toxic
Stonefly	*Pteronarcys californica*	50.0	7.0	Moderately toxic
Scud	*Gammarus lacustris*	50.0	11.0	Slightly toxic
Sowbug	*Asellus brevicaudus*	50.0	35.0	Slightly toxic

Table 6. Acute toxicity of 2,6 dichlorobenzamide (BAM) to freshwater fish and invertebrates (source: RED)

Species	Scientific Name	% ai	96-h LC 50 (ppm)	Toxicity Category
Fish				
Rainbow trout	*Oncorhynchus mykiss*	97.0	235	Practically nontoxic
Rainbow trout	*Oncorhynchus mykiss*	100.0	140	Practically nontoxic
Bluegill sunfish	*Lepomis macrochirus*	100.0	120	Practically nontoxic
Guppy	*Poecilia reticulata*	97.0	275	Practically nontoxic
Invertebrates				
Waterflea	*Daphnia magna*	97.0	856	Practically nontoxic

Table 7. Acute toxicity of dichlobenil to estuarine fish and invertebrates (source: EFED Pesticide Ecotoxicity Database and RED)

Species	Scientific Name	% ai	96-h LC 50 (ppm)	Toxicity Category
Sheepshead minnow	*Cyprinodon variegatus*	98.9	14.0	Slightly toxic
Sheepshead minnow	*Cyprinodon variegatus*	98.8	12.7	Slightly toxic
Pink shrimp	*Panaeus duorarum*	98.9	> 1.0	Moderately toxic
Mysid	*Mysidopsis bahia*	98.8	2.35	Moderately toxic
Eastern oyster	*Crassostrea virginica*	98.8	1.63	Moderately toxic
Eastern oyster	*Crassostrea virginica*	98.9	2.5	Moderately toxic

Adverse chronic effects on reproduction or growth of freshwater fish and invertebrates occurred at exposure concentrations of 0.33 ppm of technical dichlobenil for fish and 1.0 ppm for invertebrates. The degradate, BAM, is less toxic than the parent as chronic effects in fish did not occur until concentrations in the water were18 ppm, and the invertebrates were not affected at the highest test concentration of 320 ppm.

Table 8. Chronic toxicity of dichlobenil to fish and invertebrates (source: EFED Pesticide Ecotoxicity Database and RED)

Species	Scientific Name	% ai	Duration	Endpoints affected	NOEC (ppm)	LOEC (ppm)
Technical Dichlobenil						
Rainbow trout	*Oncorhynchus mykiss*	99.4	60 days	mortality	0.66	1.2
Rainbow trout	*Oncorhynchus mykiss*	99.4	60 days	length	<0.33	0.33
Waterflea	*Daphnia magna*	99.0	21 days	delayed reproduction	0.56	1.0
BAM						
Rainbow trout	*Oncorhynchus mykiss*	97.0	60 days	60-d survival, length and weight	10.0	18.0
Waterflea	*Daphnia magna*	97.0	21 days	reproduction and 21-day survival	>320	N/A

OPP does not categorize toxicity to plants. However, the data indicate that dichlobenil is more toxic to aquatic vascular plants than to algae.

Table 9. Acute toxicity of dichlobenil to aquatic plants (source: EFED Pesticide Ecotoxicity Database and RED)

Species	Scientific Name	% ai	NOEC (ppm)	EC50 (ppm)
Duckweed	*Lemna gibba*	99.0	0.006	0.030 (14-D)
Freshwater diatom	*Navicula pelliculosa*	99.0	0.31	1.00 (5-D)
Green algae	*Selenastrum capricornutum*	99.0	0.16	1.50 (5-D)
Blue-green algae	*Anabaena flos-aquae*	99.0	2.5	2.90 (5-D)
Marine diatom	*Skeletonema costatum*	99.0	0.63	2.10 (5-D)
Estuarine algae	*Isochrysis galbana*	98.9	No data	60.0 (240-h)
Estuarine algae	*Dunaliella tertiolecta*	98.9	No data	60.0 (240-h)
Estuarine algae	*Phaeodactylum tricornutum*	98.9	No data	25.0 (240-h)
Estuarine algae	*Chlorella pyreoidosa*	98.9	No data	100.0 (240-h)
Estuarine algae	*Chlorococcum* sp.	98.9	No data	60.0 (240-h)

There are some aquatic toxicity data for dichlobenil from EPA's AQUIRE database (http://www.epa.gov/ecotox/). We did not look at the original papers but report the toxicity values for the toxicity test periods that are analogous to the those required by OPP testing requirements as a means of comparison. The AQUIRE reference numbers for each reported value are provided. The data corroborate the toxicity values reported in EFED's database and the dichlobenil RED and even indicate that dichlobenil in a number of test results has lower toxicity than reported by OPP. The range of acute toxicity values for the active ingredient from AQUIRE are 10.9 to 1407 ppm for freshwater fish and 7.4 to >100 ppm for freshwater invertebrates compared to 4.93 to 12.6 ppm and 6.2 to 34 ppm for fish and invertebrates, respectively, from OPP data. Most of the data in AQUIRE are reported from studies conducted with formulated products, however, the types of formulations and percents active ingredient were not reported. Therefore, it is difficult to compare these data with those reported by OPP.

Table 10. Summary of acute toxicity data from the EPA AQUIRE database.

Species	Scientific Name	Test Chemical*	96-h LC 50 (ppm)	Reference
Freshwater Fish				
Grass carp, white amur	*Ctenopharyngodon idella*	Form.	9.4	575
Common, mirror, colored carp	*Cyprinus carpio*	Active	10.9	18518
Zebra danio	*Danio rerio*	Form.	16.0	10371
Bluegill sunfish	*Lepomis macrochirus*	Form.	10.0 - 11.0	2135, 2871
Bluegill sunfish	*Lepomis macrochirus*	Active	1407	2598
Largemouth bass	*Micropterus salmoides*	Active	12.5	2598
Rainbow trout	*Oncorhynchus mykiss*	Form.	18.0	2871
Red rasbora (harlequin fish)	*Rasbora heteromorpha*	Form.	3.5 - 4.2	848
Roach	*Rutilis rutilis*	Form.	9.5 (48-h)	8736
Medaka	*Oryzias latipes*	Form.	>40.0 (48-h)	15192
Estuarine Fish				
Striped bass	*Morone saxatilis*	Form.	6200.0	966
Freshwater Invertebrates				
Mayfly	*Callibaetis*	Active	7.4, 10.3	893
Midge family	Chironomidae	Active	7.8	893
Waterflea	*Daphnia magna*	Form.	6.2, 10.0 (48-h)	344, 886,
Waterflea	*Daphnia pulex*	Form.	3.7 (48-h)	888, 10337
Damselfly	*Enallagma*	Active	20.7	893
Scud	*Gammarus fasciatus*	Form.	10.0	886
Scud	*Hyallela azteca*	Active	2.8, 8.5	893
Dragonfly	*Libellula*	Form.	>100	893
Dragonfly	*Libellula*	Active	>100	893
Caddisfly	*Limnephilus*	Active	12.0 - 28.5, >45.0	893
Crayfish	*Orconectes nais*	Form.	22.0 (48-h)	886

Stonefly	*Pteronarcys californicus*	Form.	6.6	2871
Freshwater Plants				
Alligator weed	*Alternanthera hiloxeroides*	Form.	1.0 (24-h)	8714
Green algae	*Scenedesmus abundans*	Form.	2.7	1167
Estuarine Invertebrates				
Harpacticoid copepod	*Nitocra spinipes*	Form.	0.270	5185
Opossum shrimp	*Americamysis bahia*	Form.	2.35	344
Sand shrimp	*Crangon crangon*	Form.	3.3 - 10.0 (48-h)	906
Green crab	*Carcinus maenas*	Form.	10.0	906
Cockle	*Cerastoderma edule*	Form.	>100.0	906

* Form. = Test was conducted with formulated products. The product composition and percent active ingredient were not given.
Active = Test was conducted with the active ingredient, but the percent dichlobenil was not given.

The AQUIRE database is not always reliable regarding the test being with the formulation or the active ingredient; unless the test indicates an active ingredient, it is inputted into AQUIRE as formulation testing. However, we have seen values reported for the technical material in Mayer & Ellersieck (1986) to be reported in AQUIRE as a formulation test We report the information on formulation versus active ingredient, but we need to note that it is not completely reliable.

b. Environmental fate and transport

(The information in this section is condensed from the 1998 RED, pages 53 to 59 and 62 to 63.)

Dichlobenil dissipates in the environment on soil and in surface waters principally through volatilization. However, it is persistent under field conditions that reduce the potential for volatilization, such as cooler climates. Terrestrial and aquatic field dissipation studies showed longer half-lives in the studies conducted in Oregon (241 days 69 days, respectively) compared to those conducted in warmer climates (16 and 15 days, respectively). It is not subject to photodegradation. When it is transformed through aerobic soil metabolism, the metabolite, 2,6-dichlorobenzamide (BAM) is generated (13.1% at 50 weeks). In the soil aerobic metabolism study, the calculated half-life was 13 weeks, and 46.3 weeks when effects from volatilization were corrected. Under conditions where dichlobenil does not volatilize there is potential for both dichlobenil and BAM to move to ground water in coarse-textured soils low in organic matter. Both compounds can be extremely mobile and persistent under anaerobic conditions. Dichlobenil and BAM exceed levels of concern for ground water quality. Dichlobenil is predicted to volatilize from most surface waters. Dichlobenil is transported into surface water in the dissolved phase of the runoff water as opposed to that adsorbed onto eroding soil or entrained in sediment, based on its low to intermediate soil/water partitioning coefficients. Its persistence in surface waters will depend primarily on the environmental factors, such as temperature, which control rates of volatility. There are insufficient data to assess the persistence of BAM in surface water.

Dichlobenil does not accumulate significantly in fish in laboratory studies. BCFs were 32X in fillets, 63X in whole fish and 110X in viscera. Depuration ranged from 85% to 89% after one day of the 14-day depuration period. When a 10G formulation of dichlobenil was applied to a pond at a rate of 15 lb ai/acre, accumulation ranged from 10X to 35X in the fillets and viscera of trout, bass and catfish in the pond. The degradate, BAM, was detected in whole fish and viscera at maximum concentrations ranging from 0.013 ppm to 0.028 ppm.

c. Incidents

OPP maintains two databases of reported incidents. The Ecological Incident Information System (EIIS) contains information on environmental incidents which are provided voluntarily to OPP by state and federal agencies and others. There have been periodic solicitations for such information to the states and the U. S. Fish and Wildlife Service. The second database is a compilation of incident information known to pesticide registrants and any data conducted by them that shows results differing from those contained in studies provided to support registration. These data and studies (together termed incidents) are required to be submitted to OPP under regulations implementing FIFRA section 6(a)(2).

We are aware of only two incident reports for dichlobenil, and both involved terrestrial plants in Washington State.

d. Estimated and measured concentrations of dichlobenil in surface waters

Estimated environmental concentrations (EECs)

In the environmental risk assessment in the 1998 RED, OPP's Environmental Fate and Effects Division (EFED) derived aquatic EECs from Tier 1 modeling, the GENEEC model. This model is based on a high runoff scenario to vulnerable soils as modeled on a site in Mississippi, and it calculates generic EECs based on runoff from a ten-hectare field to a one-hectare pond, 2 meters deep. The model is designed to take into account degradation of the active ingredient in the field prior to a rain event, and assumes that off-site drift is minimal from granular formulations. The EECs as reported in the RED are presented in Table 11. Note, that OPP does not have valid scenarios for homeowner uses or noncrop areas such as rights-of-way. Therefore, although the rates listed in the table include those used on homeowner and noncrop sites, the EECs do not realistically represent the levels of dichlobenil entering surface waters from these use sites.

Table 11. Estimated Environmental Concentrations (EECs) for Aquatic Exposure Modeled with GENEEC

Rate (lb a.i./a)	Peak EEC (ppb)	Day 4 EEC (ppb)	Day 21 EEC (ppb)	Day 56 EEC (ppb)
Not soil incorporated				
2	95	65	18	7
4	190	129	36	14
6	285	194	54	20
8	380	259	72	27
20	951	648	180	67
2 **2 inch soil incorporation**				
2	47	32	9	3
4	95	65	18	7
6	141	96	27	9
8	190	130	36	14
20	476	324	90	34

The residue values in the table above indicate that for the simulations without soil incorporation the range of peak aquatic EECs was 95 ppb for 2 pounds a.i./a to a maximum of 951 ppb at 20 pounds a.i./a. The peak EEC varied by a factor of 47.5 ppb for each pound increase in the application rate of dichlobenil. For the model results with soil incorporation to a 2-inch depth, the maximum EEC was estimated to be 476 ppb, and the peak EEC changed by 23.8 ppb for each pound of dichlobenil applied. Using this factor of 23.8, we can estimate that an application of 10 pounds a.i./a, followed by a 2-inch incorporation would be approximately 238 ppb.

Comparisons of the modeling results between unincorporated and soil-incorporation to a depth of 2 inches suggests that aquatic EECs for dichlobenil are reduced approximately one-half following this incorporation method. As the label actually requires incorporation to a depth of 4 to 6 inches for the application rate of 10 to 20 pounds a.i./a for nutsedge control on noncrop areas, we ran the GENEEC program using the same input parameters presented in the RED for a 4-inch soil incorporation at 20 pounds a.i./a. The peak and 4-day EECs were 257.5 ppb and 206 ppb, respectively. The new label that is under review in OPP requires a maximum rate of 10 pounds a.i./a with a 4 to 6 inch incorporation. An estimated peak EEC for this rate is approximately 130 ppb. This indicates that the deep incorporation required for the high application rates essentially reduces peak EEC concentrations to a rate equivalent to no more than 6 pounds a.i./a unincorporated. Finally, the 20 pound unincorporated rate was modeled to simulate homeowner treatment of driveways and walkways. As discussed in the section on rates and uses, this rate is no longer registered.

As discussed in the RED these EECs are likely to be higher than we would actually expect in California and the Pacific Northwest (PNW). Dichlobenil is highly volatile, and GENEEC does not directly incorporate volatilization in the model. Although the laboratory metabolism half-lives used in GENEEC partially reflect losses due to volatilization, they do not completely account for this process. Dichlobenil primarily dissipates via volatilization, therefore, the aquatic EECs reported for dichlobenil are considered to be maximum values since this route of dissipation was not completely reflected in the input parameters for the model. Also, assumptions made in GENEEC concerning losses due to runoff are very conservative and likely overestimate the concentrations of dichlobenil in surface waters.

Another aspect that contributes to the conservativism of the GENEEC EECs with respect to the concentrations of dichlobenil likely to be present in surface waters of California and the PNW is the use of dichlobenil in this region compared to the field parameters of the model. The model assumes that a chemical is applied to large agricultural fields (10 hectares = 24.7 acres), but the use information we have from the several sources discussed earlier all indicate that the agricultural uses are a minor portion of the amounts of dichlobenil used in these states. When dichlobenil is applied to agricultural fields in this region, the applications are localized and generally spot treatments, and the total poundage applied statewide in each of the four states is relatively small. The noncrop and homeowner applications, the breakdowns of which are generally unspecified, except for the California use information as shown in Table 3, constitute the largest uses by amounts of active ingredient applied. However, the amount of each site that is treated with dichlobenil is usually not specified. Nevertheless, as we know dichlobenil is applied as spot treatments particularly around homes or in relatively small swaths for noncrop uses, it is our professional judgement that the EECs overestimate the amount of dichlobenil that is transported into waterways from these groups of use sites.

Furthermore, the loss of dichlobenil through volatilization is significant when stream dynamics are added. According to an OPP/EFED environmental fate scientist, the conditions of moderate to high volatility, moderate solubility, partitioning into the water phase during off-site transport and the dynamics of moving water indicate the EEC values for dichlobenil are overestimations of what is likely occurring in the stream habitats of salmonids in the PNW.

The degradate BAM, cannot currently be modeled in surface water as additional data on its environmental fate are needed. These studies were required in the 1988 RED, and have been submitted by the registrant, but have yet been reviewed. However, the toxicity data indicates that BAM is practically nontoxic to aquatic organisms.

Measured concentrations in water

At the time the RED was written there were only limited surface water monitoring data available. The RED included a summary of dichlobenil detections in the STORET database. The maximum concentration of dichlobenil was 0.32 ppb for a filtered surface water sample collected from the South Platte River in Colorado in 1993 to 1994. The range of concentrations in filtered water samples was 0.04 to 0.32 ppb.

We requested, and Crompton Corp. submitted, monitoring data for dichlobenil in California and the PNW states. The registrant submitted USGS NAWQA water monitoring data for the years 1993 to 2000 for concentrations of dichlobenil in surface and ground waters. There were a total of 1592 samples, 850 surface water and 742 ground water, from sites in California, Idaho, Oregon and Washington. The limits of detection ranged from 0.02 to 0.12 ppb, but was generally 0.02 ppb. Thirty-six samples had estimated detections ranging from 0.0003 ppb to 1.2 ppb. The median value was 0.04 ppb, and the seven highest detections ranged from 0.12 ppb to 1.2 ppb. The information from the NAWQA data relevant to the salmonid ESUs are summarized in Table 12. The estimated detections are shaded in the table.

Table 12. Counties containing the spawning and rearing habits of the Steelhead ESU:

State	County	Ranges detected ppb
Ca	Butte	Ag <.02
		Mixed <.02
Ca	Colusa	Ag. < .02
Ca	Glenn	Ag < .02
Ca	Merced	Mixed <0.04 - .02 Est. = .09
		Ag < .02
		Other < .02

Ca	Nevada	Other < .02
Ca	Placer	Mixed < .02
Ca	Sacramento	Urban < 1.2-.02
		Ag < .02
Ca	San Joaquin	AG<.05-.02
		Mixed<.02
Ca	Stanislaus	Ag<.02
		Mixed<.09 - .02
Ca	Sutter	ag<.02
		mixed<.02
Ca	Yolo	ag< 1.2-.02
		mixed<.02
Ca	Yuba	mixed<.02
Id	Blaine	ag<.02
		mixed<.02
Id	Custer	other<.02
		mixed<.02
Id	Jefferson**	mixed<.02
Id	Latah	ag<1.2
		other<1.2
Or	Benton	ag<.02
		mixed<.02
		other<.02
Or	Clackamas	ag<.02
		Mixed<.02
Or	Lane	mixed<.02
		other<.02
Or	Linn	ag<.02

		Mixed<.02
Or	Marion	
		mixed<.02
		other<.02
Or	Multnomah	mixed<.05- .02
		urban<.02
		other<.02
Or	Polk	Mixed<.02
Or	Washington	mixed<.02
		other<.02
Or	Yamhill	mixed<.02
		ag<.02
Wa	Adams	mixed<1.2
		ag<1.2
		other<1.2
Wa	Douglas	mixed<1.2
Wa	Franklin	mixed<1.2
		ag<1.2
		other<1.2
Wa	Grant	ag<1.2
		mixed<1.2
		other<1.2
Wa	King	
Wa	Kitsap	mixed<1.2
Wa	Lincoln	ag<1.2
Wa	Mason	mixed<1.2

Wa	Pierce	Mixed<1.2
		Urban<1.2
Wa	Snohomish	Mixed<1.2
Wa	Spokane	mixed<1.2
Wa	Thurston	mixed<1.2
		urban<1.2
Wa	Whatcom	ag<1.2 Est. = .01 - .004
		mixed<1.2
Wa	Whitman	ag<1.2
		mixed<1.2
		urban<1.2
		other<1.2

No * county contains both chinook salmon and steelhead trout.
** Counties that have chinook salmon and not steelhead trout.

Crompton Corp. also submitted monitoring data from the California Surface Water Database for the years 1996 to 1998. Seventy-five samples were collected in three locations and no dichlobenil was detected in any samples. The limits of quantification were 0.02 and 1.2 ppb. These results are not unexpected considering that California DPR reported that only 1620 to 2155 pounds dichlobenil were used statewide in the years the monitoring was conducted.

The third monitoring report that was submitted was the *Washington State Pesticide Monitoring Program: 1997 Surface Water Report*. The WSPMP monitored two urban streams biweekly from April to August. These streams receive runoff from residential yards and streets with little vegetative buffer. One stream also collected surface runoff from storm drains. Twenty-one pesticides were detected, and dichlobenil was one of the five most frequently detected herbicides, but it never exceeded water quality criteria. The presence of the frequent detections was attributed to dichlobenil availability for use by homeowners.

The concentrations of dichlobenil ranged from identified but estimated values of 0.005 ppb to 0.051 ppb. There was one “outlier” measurement of 0.12 ppb one time only, and this occurred during a rainfall event in April. However, during two other rainfalls dichlobenil concentrations were within the range of 0.009 to 0.045 ppb. Dichlobenil’s metabolite, BAM, was also detected at all seven sampling sites at concentrations ranging from 0.004 ppb to 0.14 ppb.

The WSPMP report also provided summaries of monitoring data from 1992 to 1996. Dichlobenil was detected at concentrations in several creeks ranging from 0.004 ppb to 0.21 ppb, with a median value of 0.039 ppb. The highest detections occurred in drainage ditches where the dichlobenil levels ranged from 0.087 ppb to 7.5 ppb. Out of ten reported detections in the ditches, five ranged from 1.5 ppb to 7.5 ppb. BAM was also detected at levels ranging from 0.055 ppb to 0.71 ppb.

e. General risk conclusions

Our risk conclusions are based on risk quotients (RQs) derived from the available toxicity data (Tables 4 to 10) and EECs from the GENEEC model for currently labeled rates of 2 pounds a.i./a to 20 pounds a.i./a, both with and without soil incorporation. As discussed in the section on estimated concentrations, above, the 20 pound unincorporated rate for homeowner use is no longer registered. Risks associated with the asphalt and sewer pipe uses were not considered in the RED as there is minimal exposure to nontarget organisms from the methods of applications. The RQs are presented in Table 13.

Table 13. Acute risk quotients for freshwater and estuarine fish and invertebrates and aquatic vascular plants, based on toxicity for the most sensitive species from technical grade testing of the active ingredient (Tables 4 to 10) and EECs modeled by GENEEC (Table 11)

LB ai/a	Risk Quotients[6]											
	Unincorporated						2-inch Incorporation					
	Peak EEC ppb	RQ FW Fish[1]	RQ FW Inv[2]	RQ Est. Fish[3]	RQ Est. Inv[4]	RQ Plant[5]	Peak EEC ppb	RQ FW Fish[1]	RQ FW Inv[1]	RQ Est. Fish[3]	RQ Est. Inv[4]	RQ Plant[5]
2	95	0.02	0.03	0.01	0.04	3.17	47	0.01	0.02	<0.01	0.02	1.57
4	190	0.04	0.07	0.01	0.08	6.33	95	0.02	0.03	0.01	0.04	3.17
6	285	0.06	0.10	0.02	0.12	9.50	141	0.03	0.05	0.01	0.06	4.70
8	380	0.08	0.14	0.03	0.16	12.7	190	0.04	0.07	0.01	0.08	6.33
~~20[7]~~	~~951~~	~~0.19~~	~~0.34~~	~~0.07~~	~~0.40~~	~~31.7~~	~~476[8]~~	~~0.10~~	~~0.17~~	~~0.04~~	~~0.20~~	~~15.9~~
							4-inch Incorporation					
10							140	0.03	0.05	0.01	0.06	4.67
20							258	0.05	0.09	0.02	0.11	8.60

[1]Rainbow trout LC50 = 4930 ppb. The RED used the rainbow trout LC50 of 6260 ppb, but we selected the equally valid, but lower, and therefore, more sensitive, toxicity value for our analysis.
[2]Scud LC50 = 2800 ppb
[3]Sheepshead minnow LC50 = 12,700 ppb
[4]Mysid LC50 = 2350 ppb
[5]Duckweed EC50 = 30 ppb
[6]Peak EEC/LC50 or EC50; the acute LOC is >0.05 for endangered fish, >0.5 for aquatic-invertebrate populations, and >1 for aquatic-plant populations.
[7]This rate refers to the homeowner treatment of driveways and walkways, but the new label calls for the equivalence

of 8 to 10 lb/a for these uses. Therefore these RQs will not be considered in the risk assessment.
[8]Although the models were done for a 20 lb application and 2-inch incorporation, the current and proposed labels require a 4- to 6-inch incorporation for 10 to 20 lb/a rates. Therefore, we are not considering the RQS calculated for the 20 lb rate and 2-inch incorporation, but using the RQs for the 4-inch incorporation instead.

There are several factors that need to be considered before we discuss the risk quotients. As discussed in the background information, when there is a concern (the RQ analysis indicates that criteria are exceeded) a more sophisticated Tier 2 PRZM-EXAMS model is run to refine the EECs derived from the GENEEC model if a suitable scenario has been developed and validated. However, as there are currently no valid scenarios for the noncrop uses where the higher application rates are causing the higher RQS, the Tier 2 models were not run. According to an OPP/EFED environmental engineer who has developed and validated a number of surface water models, the GENEEC model, which is based on sites in Mississippi overpredicts the EECs as compared to PRZM-EXAMS as the former model is more conservative. GENEEC overpredicts to a greater extent in the PNW region than in the southeastern US as there is less total rainfall, and, therefore, less runoff in the PNW. Therefore, the EECs, and hence, the RQs, in Table 14 are higher than are likely to occur, but the magnitude of the overprediction cannot be determined.

OPP uses a RQ > 0.05 (LOC > 0.05) to indicate there is a potential acute risk to endangered aquatic species. The RED concluded that since the GENEEC model cannot fully account for the rapid volatilization of dichlobenil, a major route of dissipation, the levels of concern for acute risks to endangered fish at the 8 pounds a.i./a unincorporated and at 20 pounds a.i./a with a 2-inch incorporation would not be exceeded. Only risks at the 20 pound unincorporated rate, no longer registered, exceeded the level of concern. Based upon the properties of dichlobenil and the specifications of the GENEEC model relative to conditions in California and the PNW, we concur that there will be no direct effect to listed salmonids.

The LOC for determining indirect effects of endangered salmonids through loss of their food supply is RQ > 0.5 for acute effects on aquatic invertebrates. As the RQS for acute risks to invertebrates are less than the LOC for all application rates, we conclude that dichlobenil will have no effect on the aquatic invertebrate food supply for Pacific salmon and steelhead.

Chronic risk is determined by comparing the 56-day EEC values to the trout early-life stage NOEC of <0.33 ppm and the 21-day EEC values to the waterflea life-cycle NOEC of 0.56 ppm. The 56-day EECs ranged from 3 ppb to 67 ppb, and the 21-day EECs ranged from 9 ppb to 180 ppb. The comparisons indicate that there is minimal likelihood of chronic risk to any fish and invertebrates from dichlobenil, and no effect on those that live in flowing water.

According to http://courses.cropsci.ncsu.edu/cs414/CH_8.htm, dichlobenil can be used to control rooted aquatic weeds only in non-flowing water such as ponds, reservoirs and lakes. (Those uses are no longer registered.) It kills rooted aquatic vegetation via slow translocation from the soil through the roots and upward into the plant body through the xylem. It acts by inhibiting cell division in growing meristems. It is not absorbed into the plant from the water column. Therefore dichlobenil is more likely to have an effect on rooted aquatic vegetation than floating aquatic vegetation. It is our professional judgement that although the RQS for aquatic plants exceed the LOC of 0.5, they overestimate the risks to aquatic plants in flowing waters.

According to an OPP/EFED fate scientist, dichlobenil has a low binding potential to sediments and tends to stay in the water column. In flowing waters, there is not much opportunity for it to get into the sediments from where it can be translocated through the root system. Volatilization occurs on the treated sites, the water column and in the runoff water during runoff events. All these properties of dichlobenil indicate it is more effective on inhibiting growth of aquatic weeds in lentic systems than in lotic systems, with limited potential for it to be significantly detrimental to aquatic vegetation in salmonid habitats.

As discussed in detail above, the low total poundage of dichlobenil used in California and the PNW, its application as single spot treatment in localized areas from late fall to early spring, and the conservatism of the GENEEC model, indicate that, in our professional judgement, we do not expect the registered uses of dichlobenil to have an effect on aquatic plants in waterways receiving runoff. We conclude that dichlobenil will have no effect on Pacific salmon and steelhead from impacts on aquatic plant cover.

f. Existing protective measures

Nationally, there are no specific protective measures for endangered and threatened species beyond the generic statements on the current dichlobenil labels. As stated on product labels, it is a violation of Federal law to use a product in a manner inconsistent with its labeling. FIFRA section 3 labels for dichlobenil (e.g. Casaron 4G, EPA Registration No. 400-168) have the Environmental Hazard Statement:

> Do not contaminate water when disposing of equipment washwaters. Cover, collect or incorporate granules spilled on the soil surface. Do not apply directly to water, to areas where surface water is present or to intertidal areas below the mean high water mark.

There is also a ground water statement:

> This chemical demonstrates the properties and characteristics associated with chemicals detected in ground water. The use of this chemical in areas where soils are permeable, particularly where the water table is shallow, may result in ground water contamination.

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