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OFFICE OF
CHEMICAL SAFETY AND
POLLUTION PREVENTION

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MEMORANDUM **DP Barcode: 451205**

RE:	**1,3-dichloropropene (1,3-D):** Draft Risk Assessment (DRA) in Support of Registration Review
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The Environmental Fate and Effects Division (EFED) conducted an abbreviated, qualitative draft ecological risk assessment (DRA) in support of the registration review of 1,3-dichloropropene (1,3-D), a soil fumigant. The decision not to conduct a full quantitative risk assessment, jointly made with the Pesticide Re-evaluation Division (PRD), is based primarily on an evaluation of the data submitted since the last major risk assessment, which generally support previous risk conclusions.

1 Executive Summary

This streamlined draft ecological risk assessment (DRA) was developed in support of the registration review of 1,3-D, a soil fumigant. 1,3-D exerts toxic action on vital enzymes or enzyme systems in nematodes, the target pest. Substitution of a sulfhydryl, ammonia, or hydroxyl group with the chlorine in 1,3-D results in restriction of the enzyme function, leading to paralysis and death (Cox, 1992). The mechanism of action in other taxa (*e.g.*, plants) is not understood.

This fumigant is a Restricted Use Pesticide that can only be applied by certified pesticide applicators and is registered for preplant use on many agricultural crop soils, turf, and nursery crops (*i.e.*, ornamental, non-bearing fruit/nut trees, and forestry crops). The only registered post-plant use is for vineyards. Maximum application rates vary from 18 lbs ai/A for grapes to 841 lbs ai/A for ornamentals. The Registration Eligibility Decision (RED) mitigation included lowered maximum application rates for some uses, deletion of selected use sites, soil moisture and sealing requirements, a 300 foot buffer from occupied structures, ground water advisory, prohibition of use on shallow groundwater and vulnerable soils in northern tier states (ND, SD, WI, MN, NY, ME, NH, VT, MA, UT, MT), a 100 foot buffer between drinking water wells and treated fields, and prohibition of use in areas overlaying karst geology. Since the RED, the prohibition of 1,3-D use in areas overlying karst geology was changed to prohibit application within 100 feet of karst topographical features and the 300-foot buffer from occupied structures was changed to 100 feet from occupied structures for some 1,3-D products combined with chloropicrin. Since the time of the RED, the maximum application rate for ornamentals (the use with the highest application rate) increased from 556 lbs ai/A to 841 lbs ai/A.

This streamlined assessment evaluates previous risk conclusions, anticipated routes of exposure, and recently submitted environmental fate and toxicity data to determine the likelihood of exposure to and risk from 1,3-D and its degradation products. Additional characterization is provided in cases where past assessments did not account for an exposure route (*e.g.*, inhalation risk to birds and honeybees) or application rate (*e.g.*, the current application rate for ornamentals). New estimated environmental concentrations (EECs) were not calculated for this assessment because of known limitations in modeling 1,3-D air concentrations (inhalation exposure) and volatility (aquatic exposure from soil injection applications) that preclude meaningful results. Instead, field volatility and run-off field studies were considered the best currently available information to inform inhalation and aquatic exposure (soil injection uses), respectively. A qualitative characterization is presented in lieu of calculating risk quotients (RQs) because of uncertainty extrapolating exposure estimates of registered uses from these field studies to a wide variety of use conditions. The current assessment generally supports the overall risk conclusions of past risk assessments while also identifying additional potential concerns as discussed below.

1.1. Characterization of Risk to Aquatic Organisms

Volatilization of 1,3-D from the application site, the primary dissipation pathway, and subsequent movement via wind may result in deposition into nearby water. Leaching and run-off is also expected to move 1,3-D and its degradates to nearby water bodies. Once reaching these environments, some 1,3-D is expected to volatilize, while the remainder will likely degrade into the major degradates forming in soil and water, 3-chloroallyl alcohol and 3-chloroacrylic acid. These compounds are expected to remain in the water column as they are less volatile and more soluble than the parent. The low octanol/water partition coefficient of 1,3-D (log K_{ow} = 1.82) indicates that it is not likely to bioconcentrate in tissues of aquatic organisms. The residues of concern (ROC) for aquatic organisms are parent 1,3-D, 3-chloroallyl alcohol, and 3-chloroacrylic acid because toxicity data indicate that fish are more sensitive to 3-chloroallyl alcohol than 1,3-D and that aquatic plants, depending on the species, are more sensitive to 3-chloroallyl alcohol, 3-chloroacrylic acid, or both in comparison to 1,3-D (*see* USEPA, 2013).

For the majority of uses (soil injection), aquatic exposure to 1,3-D is estimated from the results of a run-off field study (MRID 45022301 *as cited in* USEPA, 2013) rather than modeled EECs as in previous assessments because of limitations related to accounting for volatility in surface water modeling scenarios. The run-off field study is considered more reliable than the current models for estimating 1,3-D aquatic exposure from soil injection uses because the field study inherently accounts for volatility. In contrast, the design of the run-off field study does not represent or account for un-tarped surface applied uses of 1,3-D (*i.e.*, grape[1]); therefore, the previously modeled surface water EECs continue to be the best available estimate of this use (USEPA, 2007). Although the run-off field study was based on a single application rate and set of use conditions (*i.e.*, loam soil injected with 1,3-D to 12-inches depth with an application rate of 308 lbs ai/A), the results can be extrapolated to other application rates. The maximum instantaneous run-off concentration measured in the run-off field study was 17 µg/L, which extrapolates to a concentration that is 90% and 170% higher for application rates of 581 (the most common agricultural rate) and 841 lbs ai/A (ornamentals, the maximum rate for all uses), respectively. However, it is expected that concentrations would be lower when considered over a 24-hour averaging period[2] after accounting for dissipation processes such as volatilization and degradation in the receiving waterbody. In terms of chronic exposure, the best current estimate is a simple adjustment to peak exposure values based on aerobic aquatic metabolism, which is the degradation route most significant to aquatic 1,3-D dissipation. Finally, peak degradate exposure is estimated based on peak degradate formation in submitted studies, as discussed in **Section 5**.

Previous risk assessments (*e.g.*, USEPA, 1997; USEPA, 2007; USEPA, 2008) identified a potential acute and chronic risk concern for non-listed fish and aquatic invertebrates based on exposure to the parent compound. These potential risk concerns were based on level of concern (LOC) exceedances or inference from existing data in the absence of data (*e.g.*, chronic risk to fish). There is one incident reported for a fish kill associated with 1,3-D; however, it occurred because of an accidental spill into a nearby creek. No risk concerns were identified in previous assessments for exposure to the major soil and water degradation products (3-chloroallyl alcohol and 3-chloroacrylic acid). The previous findings were based on varying methods of modeling exposure estimates, partial datasets of required toxicity studies, and application rates lower than the current maximum labeled rate. Though some specific risk conclusions change (*see* **Section 5**), the overall conclusions of past assessments are supported when considering currently registered labels, recently submitted toxicity (*see* **Section 6**) and fata data (*see* **Section 5**), and the results of the run-off field study (*see* **Section 5**).

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[1] Application to grapes at a maximum of 18 lbs ai/A is the only 1,3-D use pattern that allows surface application without requiring a tarp to contain the application. All other uses require injection to a depth of at least six inches or require a tarp, if surface applied (USEPA, 2013). In those cases, 1,3-D may be applied through buried drip (drip lines buried at least 6 inches below soil surface) irrigation equipment without a secured tarp seal. The injection point for untarped broadcast applications must be a minimum of 12 inches from the nearest final soil/air interface.

[2] Current guidance for use of the Pesticide in Water Calculator instructs acute aquatic exposure to be modeled and assessed based on 1-in-10-year 24-average concentrations.

A risk concern was not identified for non-listed aquatic plants in previous risk assessments (*e.g.* USEPA, 2007 and USEPA, 2008); however, the RED (USEPA, 1997) did not assess risk to aquatic plants because toxicity data were not available at that time. No additional toxicity data that would change the risk picture have been received since the last risk assessment. Likewise, the current understanding of exposure (*see* discussion above and **Section 5**) is consistent with the previous findings of low risk from exposure to the parent and one of the degradation products, 3-chloroallyl alcohol. However, it is now thought that exposure to 3-chloroacrylic acid, which is more toxic than the parent, is higher than previously considered and may be great enough to be of concern for drip irrigation (*i.e.*, surface or near soil surface) application to grapes. There is also reason to consider that other uses (*i.e.*, soil injection applications) may result in 3-chloroacrylic acid exposure that reaches levels of concern because exposure estimates based on the run-off study may not represent a worst case scenario and are within only 13X of exceeding the LOC for the highest labeled application rate (841 lb ai/A).[3]

1.2. Characterization of Risk to Terrestrial Organisms

1,3-D is expected to exist solely as a vapor in the ambient atmosphere due to high vapor pressure and low Koc. Inhalation of volatilized 1,3-D is assumed to be the primary exposure route for animals and is expected to be protective of any exposure and risk from other pathways (*e.g.*, incidental soil ingestion) (USEPA, 2013). For purposes of this assessment, the inhalation ROC is only parent 1,3-D for terrestrial animals because the major degradates forming in soil and water (3-chloroallyl alcohol and 3-chloroacrylic acid) are less volatile than 1,3-D (USEPA, 2013). Furthermore, although degradates that form in air differ from those that form in soil and water, it is unknown how much they form and how their toxicity differs from parent 1,3-D. Therefore, this is an uncertainty, but it is assumed that potential risk identified from exposure to 1,3-D covers risk from those degradates. Terrestrial vertebrates and invertebrates that are on the treated field or downwind of a treated field during and shortly after treatment are expected to be at the greatest exposure to and potential risk from 1,3-D. Long-term continuous exposure is not expected at a given treatment site because 1,3-D is estimated to degrade rapidly in the atmosphere (half-life = 1.0 to 1.1 days, *as estimated in* EPISuite AOP v1.91) via reaction with free-hydroxyl radical. Although a short window of exposure on a given treatment field limits the window of opportunity of exposure, it is important to note that it does not eliminate the potential of eliciting acute or chronic effects from exposure that occurs during that period.

Available field volatility studies are considered the best available information to estimate air concentrations even though they were conducted at application rates below many of the current labeled rates. The most recently submitted studies (*see* summary of MRID 45222501 *as cited in* USEPA, 2013) show greater exposure potential (4.556 mg/m^3 based on an application rate of 51 lbs ai/A) than the study available at the time of the last comprehensive quantitative assessment of 1,3-D (USEPA, 2007); therefore, it is now thought that inhalation exposure estimates are greater than previously considered. Although there are numerous factors that may impact the actual level of volatilization and air concentrations other than application rate (*e.g.*, application type, weather, soil texture, *etc.*), all things being equal, the concentration of 1,3-D in air is assumed to be directly proportional to the application rate (*discussed in* USEPA, 2014). Based on exposure estimates extrapolated from the volatility studies by linear scaling, atmospheric concentrations may approach levels of concern on or near the treatment field, especially at the higher labeled application rates (581 lbs ai/A and 841 lbs ai/A rates are about 11-16X higher than those tested in these volatility studies).

[3] The potential risk from 3-chloroacrylic acid is much greater than that of either the parent or 3-chloroallyl alcohol.

It is unlikely that there is an acute inhalation risk concern for birds, mammals, and honeybees unless they are on or near the treatment field following an application. Mammalian acute inhalation toxicity data (MRID 0032985, *cited in* USEPA, 2013[4]) indicate that air concentrations would need to exceed 1,655 mg/m^3 to reach the non-listed species LOC of 0.5, or about 360X greater than the highest six-hour mean air concentrations observed in the volatility studies. Similarly, the air concentration would need to exceed 885 mg/m^3 (*ca.* 190X greater than the highest observed air concentration) for birds based on the recently submitted acute inhalation toxicity data (MRID 50457001). Acute inhalation risk to honeybees is more uncertain. The air concentration would need to exceed 82.5 mg/m^3 before observing any mortality based on the NOAEC in the recently submitted honeybee study (MRID 49382002). This value is close to potential air concentrations on the treatment field at the higher application rates (*as discussed above* based on linear scaling). That said, as described in **Section 6**, there is a high degree of uncertainty about the exposure concentrations in the honeybee inhalation study and a reliable LC_{50} could not be determined; therefore, risk is expected to be lower if based on a reliable LC_{50} value rather than the NOAEC. Without a reliable LC_{50} value, there is a high degree of uncertainty about acute inhalation risk to honeybees.

Chronic risk to birds and honeybees is uncertain due to a lack of toxicity data. In contrast, the maximum air concentration in the volatility studies (4.556 mg/m^3 based on an application rate of 51 lbs ai/A) suggests a potential chronic risk concern for mammals when scaled to the maximum labeled application rates (581 and 841 lbs ai/A). This is because the linearly scaled air concentrations for the higher application rates are approaching the NOAEL (91 mg/m^3; MRID 00144715 and 00152848 *cited in* USEPA, 2013). On the other hand, the LOAEL (272 mg/m^3 or 60 ppm) is three times higher than the linearly scaled concentrations from the volatility studies discussed above; therefore, it is not clear if effects would be observed if exposure occurred. Furthermore, as discussed above, the exposure window is going to be relatively short given the relatively rapid dissipation observed in field volatility studies and predicted rapid degradation in the atmosphere. Although a short exposure period does not eliminate the possibility of observing chronic effects, it does limit the opportunity for potential exposure. No incidents have been reported for animals exposed to 1,3-D except for one incident with wildlife (#024986–00011), for which there are no associated details. Although reported incidents may support a potential risk concern, the lack of reported incidents does not necessarily negate a potential risk concern because ecological incidents are often not observed and may go unreported.

Exposure to terrestrial plants is assumed to be from both run-off and vapor-phase exposure (USEPA, 2013). For purposes of the current assessment, the ROC for terrestrial plants is 1,3-D for both run-off and vapor phase exposure. Terrestrial plant toxicity data for 3-chloroallyl alcohol and 3-chloroacrylic acid show no evidence of enhanced or equivalent toxicity compared to parent 1,3-D (MRID 45007502, *cited in* USEPA, 2013); however, the tested application rates of the degradates were not high enough compared to that of 1,3-D (given the study results) to make a meaningful comparison of the relative toxicities among the three compounds. In contrast, aquatic plant toxicity data suggest that both 3-chloroallyl alcohol and 3-chloroacrylic acid have the potential to be more toxic to plants (*i.e.*, aquatic and potentially terrestrial) than the parent compound, 1,3-D (*see* USEPA, 2013). Although there is some uncertainty about the relative toxicity of the degradates, it is assumed that the previously identified potential risk concern based on exposure to 1,3-D (USEPA, 2007) covers a potential risk concern for exposure to the degradates as well. The potential risk concern, previously identified for non-listed terrestrial plants after soil-surface drip applications at rates of 18 lbs ai/A and higher (USEPA, 2007), is seemingly supported by numerous incidents with terrestrial plants that have been associated with 1,3-

[4] Note: MRID was erroneously cited for this study as 42954802 in USEPA, 2013.

D. Although not assessed in the RED due to a lack of data, uses with higher application rates, injected at soil depths greater than 5 cm, are a potential risk concern as well. That said, there remains uncertainty about a risk concern for terrestrial plants because toxicity data are not available for vapor-phase exposure to terrestrial plants (vegetative vigor stage) and the current model for terrestrial plant exposure estimates does not account for volatility, which would potentially reduce the run-off exposure estimates that are the basis of the previously identified risk concern.

While exposure to non-target terrestrial plants and animals to residues that have volatilized or run-off from the treatment area is possible, methods such as tarping treated fields serve to reduce exposure to non-target organisms that inhabit or enter the treated field or adjacent areas downwind, but not necessarily to concentrations below the level of concern.

1.3. Risk Conclusions Summary

Exposure to 1,3-D is a potential acute and chronic risk concern for non-listed fish and aquatic invertebrates whereas exposure to 3-chloroacrylic acid, a degradation product, is a potential risk concern for non-listed aquatic plants. There are potential risk concerns for drip irrigation (grape) and soil injection uses. Use-specific risk conclusions are not presented for soil injection uses given the high level of uncertainty in (1) accounting for volatility in modeled exposure estimates and (2) relying on a single run-off field study for extrapolation of exposure estimates to a wide variety of use conditions. Despite the uncertainty, it is generally expected that exposure will be lower for lower application rates, greater depths of injection, and with use of techniques such as tarping.

Exposure to 1,3-D and its degradation products appears to be generally low for terrestrial vertebrates, invertebrates, and plants but may be great enough under some situations (*e.g.*, high application rates, no tarps, high temperatures) to assume that there is a potential risk concern for individuals on a treated field or in close proximity downwind of a treated field during and shortly after treatment. Use-specific risk conclusions are not presented for animals given the uncertainty in using field volatility studies for extrapolation of inhalation exposure estimates to a wide variety of use conditions. Although there is a high degree of uncertainty about exposure and toxicity to plants, several reported incidents support a potential risk concern.

2 Introduction

This DRA examines the potential ecological risks associated with labeled uses of 1,3-D on non-target organisms. This streamlined assessment uses the best available scientific information on the use, environmental fate and transport, and toxicity of 1,3-D and its degradation products. This assessment provides updates since the problem formulation and relies on characterization from previous risk assessments to the extent possible.

3 Problem Formulation Update

The purpose of the Problem Formulation is to provide the foundation for the ecological risk assessment being conducted for the labeled uses of 1,3-D. The Problem Formulation identifies the objectives for the assessment and provides a plan for analyzing the data and characterizing the ecological risk. As part of the Registration Review (RR) process, the following detailed Problem Formulation was published to the docket:

- (USEPA, 2013)[5] Problem Formulation for the Environmental Fate and Ecological Risk, Endangered Species, and Drinking Water Assessments in Support of the Registration Review of 1,3 Dichloropropene (Telone). August 27, 2013. DP Barcode 410113.

A decision was made, jointly with the Pesticide Re-evaluation Division (PRD), to not conduct a full quantitative risk assessment for 1,3-D based primarily on an evaluation of the data submitted since the last major risk assessment, which generally support previous risk conclusions. New estimated environmental concentrations (EECs) were not calculated for this assessment because of known limitations in modeling 1,3-D air concentrations (inhalation exposure) and volatility (aquatic exposure from soil injection applications) that preclude meaningful results. Instead, field volatility and run-off field studies were considered the best currently available information to inform inhalation and aquatic exposure (soil injection uses), respectively. Environmental fate and toxicity studies submitted after the problem formulation are summarized in the sections below. Information on the labelled use patterns and previously submitted fate and toxicity studies can be found in the problem formulation.

4 Previous Risk Conclusions

EFED has conducted multiple ecological risk assessments on 1,3-D that serve as a basis for this streamlined risk assessment, including:

- Reregistration Eligibility Decision (RED) (USEPA, 1997)
- Section 3 new use on grapes assessment (USEPA, 2007)
- California Red Legged Frog Endangered Species assessment (USEPA, 2008)

The EFED RED assessment (USEPA, 1997) is the most recent comprehensive risk assessment conducted for 1,3-D. However, it did not assess the currently labeled maximum application rates for 1,3-D. Previous assessments identified potential risk concerns for fish, aquatic invertebrates, and terrestrial plants. None of the previous risk assessments assessed vapor phase exposure (*i.e.,* the major exposure route of exposure for soil fumigants) to birds, terrestrial invertebrates, or terrestrial plants due to a lack of data.

Previous risk conclusions for non-listed species (*as reported in* USEPA, 2013 and modified for clarity) are summarized in **Table 4-1**.

[5] Docket No. EPA-HQ-OPP-2013-0154

Table 4-1. Summary of Non-listed Species Risk Concerns Identified for Use of 1,3-D in Previous Assessments

Birds	Mammals	Terrestrial Plants	Terrestrial Inverts	Fish	Aquatic Inverts	Aquatic Plants	Bioaccumulation	Persistence	Degradates of Concern
No[a]	No	Yes[b]	NA[c]	Yes[d]	Yes[e]	No[f]	No	No	3-chloroallyl alcohol; 3-chloroacrylic acid

[a] Inhalation data were not available to assess risk. Inhalation risk was presumed low based solely on the findings for mammals

[b] Data were not available to assess risk to terrestrial plants at the time of the RED. The grape assessment (USEPA, 2007) identified a risk concern for dicot plants inhabiting semi-aquatic areas due to run-off from treated fields. Given the relatively low application rate for grapes (18 lbs ai/A versus a maximum application rate of 581 lbs ai/A for several crops and a maximum rate of 841 lbs ai/A for ornamentals), it was concluded likely that the RED would have identified a risk concern for terrestrial plants if toxicity data had been available. Vapor phase exposure was not considered as toxicity data were not available.

[c] Inhalation data were not available to assess risk.

[d] Data were not available to assess chronic risk to fish, although a risk concern was presumed.

[e] Data were not available to assess chronic risk to aquatic invertebrates at the time of the RED.

[f] Data were not available to assess risk to aquatic plants at the time of the RED. There was no risk concern in the grape assessment (18 lbs ai/A; USEPA, 2007); however, that rate is substantially less than the maximum application rate on current labels (841 lbs ai/A).

Yes = at least one LOC has been exceeded in previous assessments

No = LOCs have not been exceeded in previous assessments

NA = RQs have not been calculated previously

5 Environmental Fate Summary

Information on the environmental fate properties and studies submitted for 1,3-D can be found in the Problem Formulation for this DRA (USEPA, 2013). The following sections discuss environmental fate data that were submitted after the Problem Formulation was published and critical, previously reviewed data that are utilized to characterize exposure.

5.1. Laboratory Data

Due to the significance of using laboratory data as the basis for estimating aquatic exposure, this section summarizes newly submitted laboratory data and discusses how previously submitted and newly submitted laboratory data impact the exposure assessment.

Aerobic Soil Metabolism (1,3-D and 3-chloroallyl alcohol)

Two additional environmental fate studies were submitted and reviewed after publication of the Problem Formulation.

The aerobic transformation of 1,3-D was studied in four European soils for up to 77 days in closed systems (MRID 49382009; Supplemental). The soils were treated at 87.8 µg/g and were viable throughout the study. It was not confirmed that aerobic conditions were maintained in the soils throughout the study. Overall material balance averaged 91% to 94% across the four soils. Individual recoveries for two of the four soil systems were outside guideline requirements (>90% and <110%).

There was a decline in recovery over time for all soils except for the sandy clay loam soil. Because a significant amount of 1,3-D was lost to volatilization, DT50 values were calculated for 1,3-D in the soil plus volatile traps. Degradation kinetics of best fit were IORE in two soils and DFOP in two soils with half-lives ranging from 39 days to 87 days. 3-chloroacrylic acid occurred at concentrations up to 37% of applied radioactivity. 3-chloroallyl alcohol occurred at concentrations up to 2% of applied radioactivity. Volatile 1,3-D were a maximum of 28% to 46% of applied radioactivity across the four soils.

The aerobic transformation of the 1,3-D degradate, 3-chloroallyl alcohol, was studied in four European soils for 21 days or 9 days (Marchem soil only) in closed systems in darkness (MRID 49382008; Supplemental). The soils were treated at 24.7-25.6 µg/g and were viable throughout the study. It was not confirmed that aerobic conditions were maintained in the soils throughout the study. Overall material balance averaged 92% to 94% across the four soils. Individual recoveries for multiple samples in all systems were outside guideline requirements (>90% and <110%). There was a decline in recovery over time for all test soils. SFO half-lives were 0.48 days in the Charentilly clay loam soil, 0.53 days in the Thessaloniki sandy silt loam soil, 0.57 days in the Cuckney sand soil, and 0.11 days in the Marcham sandy loam soil. One major nonvolatile transformation product, 3-chloroacrylic acid, was identified in the four test soils.

Table 5-1. New Environmental Fate Properties for 1,3-D

Property	Value	MRID	Study Classification
Aerobic Soil Metabolism of 3-Chloroallyl alcohol	0.1-0.6 day	49382008	Supplemental
Aerobic Soil Metabolism of 1,3-D	39-87 days*	49382009	Supplemental

*Values include volatiles

<u>Impact on Exposure Assessment</u>

Previously reviewed data are utilized to characterize chronic exposure and exposure to degradates. In terms of chronic exposure, the best current estimate is a simple adjustment to peak exposure values based on aerobic aquatic metabolism because it is the degradation route most significant to aquatic 1,3-D dissipation. The available aerobic aquatic metabolism study (MRID 44975502 *as cited in* USEPA, 2013) showed a DT50 of 3 to 7 days post-treatment and a DT90 of 14 to 21 days post-treatment. The resulting first order degradation rate (-0.1421 day^{-1}) applied over 21-day and 60-day averaging periods results in average exposures of 33% and 12% of peak values, respectively. Thus, 1,3-D run-off from a soil injected application at the maximum rate that drains to a static waterbody, exclusively fed by an 1,3-D treated area, may be expected to have a peak 1,3-D concentration of 46 µg/L, a 21-day average concentration of 15 µg/L, and a 60-day average concentration of 5.5 µg/L.

New and previously submitted laboratory data also allow for updated conclusions on 1,3-D degradate formation. The degradation route most significant to aquatic 1,3-D dissipation (aerobic aquatic metabolism) forms 3-chloroallyl alcohol at a maximum 6.4% of applied 1,3-D one day after treatment and therefore 6.4% is assumed to be the aquatic exposure to 3-chloroallyl alcohol relative to peak values of 1,3-D aquatic exposure. The 1,3-D half-life via aerobic aquatic metabolism is 5 days, while the half-life via hydrolysis at pH 7 and 20°C is longer at 14 days, while the aerobic soil metabolism half-life is longer still at 12 to 87 days indicating that aerobic aquatic metabolism is the dominate degradation pathway. The formation of 3-chloroallyl alcohol can likely be attributed to hydrolysis as formation of 11% at day 2 is similar formation seen in the aerobic aquatic metabolism study. With regard to chronic exposure, 3-chloroallyl alcohol is quickly metabolized as its aerobic aquatic metabolism half-life is 1.2 days (MRID 44975503 *as cited in* USEPA, 2013). This rapid dissipation in metabolically active waters should preclude long term exposure to the alcohol degradate. In the absence of metabolic activity, 3-chloroallyl alcohol forms at up to 77% of applied 1,3-D via hydrolysis by study termination (22 days).

For 3-chloroacrylic acid, the degradation route most significant to aquatic 1,3-D dissipation (aerobic aquatic metabolism) forms it at a maximum of 9.5% of applied 1,3-D at seven days after treatment. However, aerobic soil metabolism forms 3-chloroacrylic acid at 19% of applied after the same duration (seven days) and 37% of applied at the maximum (day 28). Therefore 37% is assumed to be the aquatic exposure to 3-chloroacrylic acid relative to peak values of 1,3-D aquatic exposure. Because this exposure value is derived from data submitted during registration review, this assumption differs and assumes higher relative exposure than the approach taken in previous assessments (10% formation via aerobic aquatic metabolism; MRID 44975503 *as cited in* USEPA, 2013). With regard to chronic exposure, the 3-chloroacrylic acid is quickly metabolized as its aerobic aquatic metabolism half-life is 3.4 days (MRID 44975504 *as cited in* USEPA, 2013). This rapid dissipation in metabolically active waters should preclude long term exposure to 3-chloroacrylic acid.

5.2. Field Studies

Run-off of Telone Following Artificial Rainfall

Due to the significance of using this study as the basis for estimating aquatic exposure, this section summarizes its design and results and discusses its impact on the exposure assessment and risk conclusions.

Run-off of 1,3-D, applied as Telone® II Soil Fumigant, under U.S. field conditions, was examined in a bare ground plot at one site in Virginia (loam soil) (MRID 45022301 *as cited in* USEPA, 2013; Supplemental). The test substance was injected once, at a 12-inch depth, at an average application rate of 308 lbs ai/A. The test application was made to three replicate test plots (separated by plywood barriers) with *ca*. 5% slope. Immediately following application, the plot was disked and packed to seal the surface. Approximately 3 days following application, run-off water was collected during a natural rainfall event and during a 2-hour simulated rainfall event conducted to coincide with the peak flux of telone.

The replicate plots received an average of 3.7 inches of simulated rainfall over a 2-hour duration (1.85 in/hr), and a combined rainfall of 5.04 inches, including the natural rainfall that was recorded, over *ca*. 10 hours (0.504 in/hr). Approximately 62% to 81% of the total rainfall was captured as run-off.

In the natural rainfall event, combined cis- and trans-telone concentrations ranged from 0.653 to 7.969 μg/L. Total combined cis- and trans-telone mass loss due to run-off was 8.25 to 14.76 mg.

In the simulated rainfall event, combined cis- and trans-telone concentrations ranged from 4.168 to 17.227 μg/L. Peak concentrations occurred early in the event for Plots 1 and 2, with the maximum concentration observed at 15 minutes after the initiation of the simulated event and declined during the event. Telone concentrations fluctuated in Plot 3, with the peak concentration occurring at 1 hour and 25 minutes after initiation of the simulated rainfall event. Total combined cis- and trans-telone mass loss due to run-off was 211.9 to 406.3 mg.

Total combined cis- and trans-telone mass loss due to run-off from both the natural and simulated rainfall events was 220.1 to 421.1 mg, which accounted for 0.0015% to 0.0028% of the total applied telone.

Impact on Exposure Assessment

Although previous assessments used the best available models and data, the surface water modeling scenario used did not parameterize soil temperature or albedo, important parameters determining the extent of volatilization for incorporated volatile pesticides. The run-off field study is considered more reliable than the current models for estimating 1,3-D aquatic exposure from soil injection uses because it inherently accounts for volatility. Exposure can be estimated for registered uses by linear scaling of the application rate used in the study to maximum labelled application rates. Estimating exposure in this manner results in exposure estimates slightly lower than previously reported modeled EECs.

Impact on Risk Conclusions

At face value, the run-off field study suggests that exposure and thus risk from soil injection uses (i.e., all uses except grape surface drip irrigation) may be lower than suggested by exposure estimates based on EFED models, which may overestimate exposure due to modeling limitations in accounting for volatility of 1,3-D injected into the soil. For example, the run-off field study results suggest that there is an acute risk concern for non-listed aquatic invertebrates only at the highest application rate (841 lb ai/A; assuming injection to 12 inches soil depth and linear scaling between application rate and exposure estimate) whereas modeling presented in previous assessments also suggests a potential risk concern for lower application rates.

However, it is important to consider that the run-off field study represents only a single method of application (e.g., depth of injection and method of incorporation), site location (e.g., soil type), and environmental conditions (e.g., weather conditions). Although the study accounts for volatilization along with all other routes of dissipation, the myriad of factors impacting dissipation may result in lower or higher exposure concentrations under different conditions than those tested. In other words, it is unknown if the exposure estimates derived from a single study cover the range of registered uses, potential use sites, and environmental conditions. Therefore, the lack of a LOC exceedance for a specific use (e.g., acute risk to aquatic invertebrates at rates lower than 841 lb ai/A) does not necessarily indicate a lack of a potential concern since the exposure estimate is based on a single study.

Groundwater Monitoring in Winter Haven, Florida

Dow AgroScience commissioned a groundwater monitoring study designed to evaluate the groundwater contamination potential of telone and its transformation products at a golf course near Winter Haven, Florida (MRID 49275501; Supplemental). Soils in these areas were reportedly comprised of Satellite sand and Immokalee sand. The initial application of telone occurred in May 2004 (monitoring wells were installed in July and August 2007) and the study was terminated in June 2010.

Eight monitoring wells, six shallow and two deep wells, were installed in the interior of the golf course *ca*. 100 feet hydraulically down-gradient from the treated areas. The monitoring wells were installed at two locations; each received three shallow wells and one deep well. Shallow wells were screened *ca*. 1-2 to 6-7 feet below the ground surface and deep wells were screened *ca*. 14-15 to 19-20 feet below the ground surface. Soil and soil pore-water were not monitored.

Telone was applied at the site for seven consecutive years beginning in May 2004 and ending in May 2010; four of those applications (2004 through 2007) were made prior to the installation of the monitoring wells. A tracer substance was not applied following test substance applications. Groundwater samples were collected from both monitoring locations on nine occasions, approximately quarterly from November 2007 through December 2009, and at the driving range location at two additional intervals, March 9 and June 7, 2010. Average annual rainfall for the study period was 54.91 inches or 115% of the historical average (reviewer-calculated).

Telone and its 3-chloroallyl alcohol degradate were not detected in groundwater samples at ≥ the LOQ (0.05 µg/L for telone, 0.10 µg/L for 3-chloroallyl alcohol) at any sampling interval. However, the stability of the analytes in water samples was not adequately determined and an independent laboratory validation was not provided. The 3-chloroacrylic acid degradate was detected above the LOQ (0.05 µg/L) in groundwater samples one time at the September 2008 sampling event. At this event, *cis*- and *trans*-3-chloroacrylic acid were detected at 0.044 µg/L and 0.073 µg/L, respectively, after a 3-day storage period; however, re-analysis of the sample showed no detectable residues of the *cis* and *trans* isomers after an 11-day storage period.

5.3. Monitoring Data

Water

The Water Quality Portal (WQP, https://www.waterqualitydata.us/) was used to search the water monitoring data for 1,3-D detections. WQP is a cooperative service sponsored by the United States Geological Survey (USGS), the Environmental Protection Agency (EPA), and the National Water Quality Monitoring Council (NWQMC). It serves data collected by over 400 State, federal, tribal, and local agencies. There were 1,008 surface water monitoring data points from 1980 to 2018 associated with 1,3-dichloropropene. Among these data, 3 samples (0.3%) were reported above detection limits. Measurable concentrations ranged from 0.2 to 1.5 µg/L and were reported at sites in the states of Louisiana (two detections) and New York. Surface water samples for 1,3-D were collected in Arizona, Delaware, Georgia, Iowa, Louisiana, Maryland, New Mexico, New York, and Virginia. Detection limits were 1.0 µg/L or less in all states except Arizona.

The Water Quality Portal was also queried for groundwater monitoring data. Of 9,800 samples, 13 quantified detections (0.1% of samples) ranged from 0.2 to 26 µg/L. Quantified detections were found in Alaska, Florida, Illinois, Louisiana, North Carolina, New Jersey, Texas, Virginia, and Wisconsin. Though not all samples used the same methods or reporting limits, most protocols reported a detection limit under 1 µg/L. Targeted groundwater monitoring studies detected the degradates 3-chloroallyl alcohol and 3-chloracrylic acid whereas the non-targeted monitoring data did not include these chemicals as analytes, so it is unknown if they were present in those samples.

Air

The ambient air monitoring effort by California Department of Pesticide Regulation (CDPR, 2018) on 1,3-D shows highest 1-day concentrations of 5.0 ppb (6 µg/m^3) in Santa Maria, 2.8 ppb (3.4 µg/m^3) in Watsonville, 8.7 ppb (10 µg/m^3) in Oxnard, and 3.1 ppb (3.7 µg/m^3) in Camarillo. Sampling in Santa Maria and Camarillo started August 2010 whereas sampling in Oxnard and Watsonville started in October and November 2011, respectively.

In 2011, CDPR implemented an Air Monitoring Network (AMN) to weekly measure 32 pesticides, including 1,3-D, in three agricultural communities (Ripon, Salinas, and Shafter). CDPR published a report on the relationship between measured ambient 1,3-D concentrations and the reported use of 1,3-D in each of the three AMN sampling locations during the period from 2011 to 2014 (CDPR, 2018). Results showed that all three AMN sites possess unique detection-use profiles. Data indicate that the greatest number of detections/quantified detections occurred in Shafter (84%/27%) followed by Ripon (54%/18%). Detections often coincided with 1,3-D use except in late 2012 and early 2013 in Ripon when use was prevalent, but no quantifiable 1,3-D concentrations were reported. However, reporting limits decreased from 0.227 $\mu g/m^3$ to 0.027 $\mu g/m^3$ in October 2013. The highest 24-hour and 4-week exposure measurements were 45 $\mu g/m^3$ and 18 $\mu g/m^3$ (Shafter County).

Environmental Fate Conclusions

Environmental fate data submitted since the 1,3-D Problem Formulation provide additional evidence that volatility is the primary route of dissipation for 1,3-D. However, groundwater and surface water monitoring data indicate dissipation can also occur via leaching and run-off. While 1,3-D and degradates have been identified in groundwater, the detections are infrequent and found near use sites. Likewise, ambient air monitoring estimates provided in this update are associated with 1,3-D use.

6 Toxicity Summary

Information about toxicity data previously submitted for 1,3-D can be found in the 1,3-D Problem Formulation (USEPA, 2013). The following section discusses only those toxicity data that were submitted after the Problem Formulation was published. Additionally, chronic toxicity data are not available for estuarine-marine fish or the most acutely sensitive freshwater fish (walleye). However, chronic endpoints were estimated by acute-to-chronic ratio (ACR) for freshwater and estuarine fish based on the recently submitted data.

An open literature review was conducted using the ECOTOX database (last updated 2017; USEPA, 2017). There are no toxicity studies in the database that show greater toxicity than available studies.

6.1. Toxicity Data

Tables 6-1 and **6-2** summarize the toxicity endpoints for data submitted subsequent to the registration review problem formulation (USEPA, 2013) and includes ACR-estimated NOAEC values for chronic toxicity to fish using the recently submitted data.

Table 6-1. New Aquatic Toxicity Endpoints

Taxonomic Group	Study Type	Test Substance (% ai)	Test Species	Toxicity Value (all units in terms of µg/L measured ai)	MRID Classification	Comments
1,3-dichloropropene (parent)						
Freshwater fish	Acute (96-hr)	TGAI (100)	Rainbow trout (*Oncorhynchus mykiss*)	LC_{50} = 2780 NOAEC = 1460 LOAEC = 2130 based on sublethal effects (lethargy, partial loss of equilibrium, and erratic swimming)	49382003 Acceptable	Endpoints affected: mortality and sublethal effects (lethargy, immobility, partial/complete loss of equilibrium, erratic swimming, and swimming at surface) NOAEC/LOAEC visually determined
	Chronic (ELS)	TGAI (96.8)	Fathead Minnow (*Pimephales promelas*)	NOAEC = 8.7 LOAEC = 15 based on growth (reduced dry weight)	49682401 Acceptable	Endpoints affected: larval survival (post-hatch success), length, and weight (dry and wet) Dry weight reduced ≥ 10%
	Chronic	NA	Walleye (*Stizostedion vitreum*)	NOAEC = 2.3 LOAEC = 3.9	NA	Walleye is the most acutely sensitive species tested. Estimated by ACR using existing fathead minnow and walleye data[1]
Estuarine-marine fish	Chronic	NA	Sheepshead minnow (*Cyprinodon variegatus*)	NOAEC = 1.8 LOAEC = 3.2	NA	Chronic toxicity study unavailable. Estimated by ACR using existing fathead minnow and sheepshead minnow data[1]
3-Chloroacrylic acid (degradate)						
Freshwater fish	Chronic (ELS)	TGAI (100)	Fathead Minnow (*Pimephales promelas*)	NOAEC ≥ 9910 LOAEC > 9910 No effects	49382004 Acceptable	

Taxonomic Group	Study Type	Test Substance (% ai)	Test Species	Toxicity Value (all units in terms of µg/L measured ai)	MRID Classification	Comments
Freshwater invertebrates	Chronic (LC)	TGAI (100)	Water Flea (*Daphnia magna*)	NOAEC = 2530 LOAEC = 5080 based on growth (reduced length)	49382005 Acceptable	Endpoints affected: length reduced ≥ 7%

TGAI=Technical Grade Active Ingredient; ai=active ingredient
NA = not applicable
ELS: early-life stage; LC: life cycle
[1] Fathead minnow acute toxicity (LC_{50} = 4100 µg ai/L; MRID 40098001 *as cited in* USEPA, 2013); Fathead minnow chronic toxicity (NOAEC = 8.7 µg ai/L and LOAEC = 15 µg ai/L; MRID 49682401); Walleye acute toxicity (LC_{50} = 1080 µg ai/L; MRID 40098001 *as cited in* USEPA, 2013); Sheepshead minnow acute toxicity (LC_{50} = 870 µg ai/L; MRID 44843901 *as cited in* USEPA, 2013 . Fathead minnow ACR = fathead minnow LC_{50}/ fathead minnow NOAEC or LOAEC. Walleye ACR-estimated NOAEC or LOAEC = walleye LC_{50}/fathead minnow ACR. Sheepshead minnow ACR-estimated NOAEC or LOAEC = sheepshead minnow LC_{50}/fathead minnow ACR.

Table 6-2. New Terrestrial Toxicity Endpoints

Taxonomic Group	Study Type	Test Substance (% ai)	Surrogate Species	Toxicity Value (all units in terms of measured ai)	MRID Classification	Comments
1,3-dichloropropene (parent)						
Birds	Acute Dietary (3-day exposure; 5-day post-exposure)	Telone II microcapsule (microcapsule contained 25.6% of Telone II (96% ai))	Bobwhite quail (*Colinus virginianus*)	LC_{50} > 5970 mg/kg diet NOAEC = 776 mg/kg diet LOAEC = 1086 mg/kg diet based on growth (reduced body weight gain during the exposure period)	49382006 Acceptable	Endpoints affected: reduced growth (body weight gain) No treatment related mortality NOAEC/LOAEC visually determined
			Mallard duck (*Anas platyrhynchos*)	LC_{50} > 5980 mg/kg diet NOAEC = 369 mg/kg diet LOAEC = 691 mg/kg diet based on growth (reduced body weight gain during the post-exposure period)	49382007 Acceptable	Endpoints affected: mortality, reduced food consumption, and reduced growth (body weight gain) NOAEC/LOAEC visually determined

Taxonomic Group	Study Type	Test Substance (% ai)	Surrogate Species	Toxicity Value (all units in terms of measured ai)	MRID Classification	Comments
Birds	Acute Inhalation (4-hr whole body exposure; 14-day post-exposure)	TGAI (97.5)	Bobwhite Quail (*Colinus virginianus*)	LC_{50} = 390 (359-423)[1] ppm [*LC_{50} = 1770 (1630-1920) mg/m^3*] NOAEC = 105 ppm [*480 mg/m^3*] LOAEC = 199 ppm [*900 mg/m^3*] based on partial eye closure	50457001 Acceptable	Endpoints affected: mortality and sublethal effects (eye closure – partial or complete, gasping, cool body, hypoactivity, prostration, labored respiration, salivation, and rales – abnormal lung sounds) NOAEC/LOAEC visually determined
Terrestrial invertebrates	Acute Inhalation Toxicity (6-hr whole body exposure; 48-hr observation period)	TGAI (95.9)	Honey bee (*Apis mellifera*)	NOAEC = 18.2 ppm [*82.5 mg/m^3*]	49382002 Supplemental For qualitative use only	Although mortality ranged from 0% to 100% in a dose responsive manner, a reliable LC_{50} value cannot be calculated due to high variability in measured test concentrations. NOAEC visually determined

TGAI=Technical Grade Active Ingredient; ai=active ingredient
[1] (95% confidence interval)

1,3-Dichloropropene (1,3-D)

Six additional toxicity studies were submitted and reviewed after the release of the problem formulation.

The results of the acute fish study with rainbow trout (LC_{50} = 2780 µg ai/L; MRID 49382003; Acceptable) are consistent with the existing acute toxicity to fish data set. Walleye remains the most acutely sensitive species tested (LC_{50} = 1080 µg ai/L; USEPA, 2013; MRID 40098001).

1,3-D (Telone XRM-5048 TGAI) inhibited the growth of fathead minnow in an early life stage (ELS) study (NOAEC = 8.7 µg ai/L and LOAEC = 15 µg ai/L; MRID 49682401; Acceptable). Dry weight was reduced by 10% compared to the negative control at the LOAEC. Larval survival (post-hatch success), length, and wet weight were inhibited at higher concentrations.

Chronic toxicity data are not available for estuarine-marine fish or the most acutely sensitive freshwater fish (walleye). However, an ACR can be used to estimate chronic NOAEC and LOAEC values for those species based on the preexisting acute toxicity study for fathead minnow (MRID 40098001 *as cited in* USEPA, 2013) and the recently submitted ELS toxicity study for fathead minnow (MRID 49682401). Based on the fathead minnow ACR (*see* **Table 6-1** for details), the ACR-estimated NOAEC and LOAEC for walleye are 2.3 and 3.9 µg ai/L. Similarly, the ACR-estimated NOAEC and LOAEC for sheepshead minnow are 1.8 and 3.2 µg ai/L.

Telone (microencapsulated product) is practically non-toxic to Bobwhite quail (LC_{50} > 5970 mg ai/kg-diet, MRID 49382006; Acceptable) and Mallard duck (LC_{50} > 5980 mg ai/kg-diet, MRID 49382007; Acceptable) on an acute dietary basis. These findings are consistent with the existing 1,3-D toxicity data, which showed no mortality up to 10,000 mg/kg-diet (Telone II) in the same species (USEPA, 2013; MRID 00120907 and 00120908). Although there were a few mortalities in the recently submitted studies, there is no indication that they were necessarily treatment-related. All four of the available studies showed decreased body weight gain in Bobwhite quail and Mallard duck and decreased food consumption in Mallard duck.

A 4-hour whole-body exposure to 1,3-D vapor resulted in an LC_{50} = 390 ppm (1.77 mg/L) after the subsequent 14-day observation period (MRID 50457001; Acceptable). Several sublethal effects were observed at concentrations of 199 ppm (0.9 mg/L) and higher including eye closure – partial or complete, gasping, cool body, hypoactivity, prostration, labored respiration, salivation, and rales – abnormal lung sounds.

Adult honeybees were exposed to 1,3-D (Telone II TGAI) vapor for 6-hours and were observed for 48-hours (MRID 49382002; Supplemental, qualitative use only). There was a clear dose response that ranged from 0% treatment-related mortality at the lowest treatment level to 100% mortality at the highest treatment level. However, there was high, unexplained variability in the measured exposure concentrations over time, among treatment levels, and among bee vs. no bee enclosures. This variability also resulted in overlapping exposure concentrations among treatment levels. Given the high level of uncertainty, a reliable LC_{50} value cannot be calculated from the measured test concentrations. Furthermore, nominal test concentrations may not provide an appropriate estimate given that measured concentrations were a small fraction of the nominal concentrations. Nonetheless, given the clear dose response with increasing nominal concentration, the measured concentration at the treatment level showing no mortality (NOAEC = 82.5 mg/m^3 [18.2 ppm] based on the time-weighted average over the 6-hour exposure) can be used as a conservative endpoint for this study.

3-Chloroallyl alcohol (degradate)

The recently submitted 1,3-D acute toxicity to rainbow trout study (*as discussed above*) is consistent with the preexisting rainbow trout studies that indicate 3-chloroallyl alcohol is slightly more toxic than 1,3-D to rainbow trout on an acute basis (USEPA, 2013). However, the difference in parent to degradate toxicity is less based on the recently submitted data (3-chloroallyl alcohol is 2.8X more toxic), which show greater acute toxicity of 1,3-D to rainbow trout than the preexisting rainbow trout data.[6] 3-chloroallyl alcohol toxicity data are not available for the test species most sensitive to 1,3-D (freshwater: walleye; estuarine marine: sheepshead minnow). However, it is anticipated that 3-chloroallyl alcohol may be about 2.8X more toxic to those test species than 1,3-D if the 1,3-D to 3-chloroallyl alcohol ratio of acute toxicity to rainbow trout is conserved across species and from acute to chronic toxicity.

3-Chloroacrylic acid (degradate)

Two additional chronic toxicity studies (freshwater fish and invertebrates) were submitted and reviewed after the release of the problem formulation. These results are consistent with available data that show reduced acute toxicity of 3-chloroacylic acid to freshwater fish and invertebrates in comparison to that of parent 1,3-D (USEPA, 2013).

3-chloroacrylic acid had no treatment-related effects on fathead minnow in an early life stage study up to a concentration of 9910 µg ai/L (NOAEC ≥ 9910 µg ai/L; MRID 49382004; Acceptable). In comparison, 1,3-D affected fathead minnow at concentrations of 15 µg ai/L and higher (NOAEC = 8.7 µg ai/L; MRID 49682401).

Daphnid growth was inhibited by exposure to 3-chloroacrylic acid in a life-cycle study (MRID 49382005; Acceptable). The NOAEC = 2530 µg ai/L and the LOAEC = 5080 µg ai/L based on reduced length (≥ 7%) compared to the control. In comparison, 1,3-D affected Daphnid at concentrations of 105 µg ai/L and higher (NOAEC = 70 µg ai/L; MRID 45007501). Reduced length and progeny were the most sensitive endpoints from exposure to 1,3-D.

Toxicity Conclusions

The toxicity data submitted since the Problem Formulation was published provide additional evidence that 1,3-D and its degradation product, 3-chloroacrylic acid, are toxic to aquatic and terrestrial taxa and support the risk conclusions developed in previous risk assessments. Consistent with previously submitted data, the recently submitted data indicate that the degradate, 3-chloroacrylic acid, is less toxic than the parent compound to fish and aquatic invertebrates.

6.2. Incident Data

EFED conducted a search of the OPP Incident Data System (IDS) on March 12, 2019. Four new incidents have been attributed to 1,3-D use since the Problem Formulation was published in 2013 (USEPA, 2013). The problem formulation reported incidents with terrestrial plants (13), aquatic plants (1), and wildlife (1). All plant incidents in the EIIS database were attributed to 1,3-D applications with certainties ranging from “possible” to “highly probable”. Certainty of a causal relationship between 1,3-D and the reported incident is not included for the wildlife incident or 5 of the 13 plant incidents, all reported in the Aggregate Incident Database, which is a database that does not include incident details.

Registrants reported three new minor plant incidents between 2017 and 2018 in the aggregate incident reports. No additional details are available for these incidents.

[6] Existing 1,3-D acute toxicity to rainbow trout LC_{50} = 3.94 µg ai/L (MRID 00039692) and LC_{50} = 5.9 µg ai/L (MRID STEOD101). Existing 3-chloroallyl alcohol acute toxicity to rainbow trout LC_{50} = 0.986 µg ai/L (MRID 44940306). All as cited in USEPA, 2013.

The new terrestrial plant incident (#I029870-0007) reported in EIIS database occurred in 2017. A tomato crop was treated on several farms with Telone EC in Lazio, Italy. Transplanted seedlings were affected after the subsequent planting cycle. The certainty that this incident is attributed to Telone EC is classified as “possible”.

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