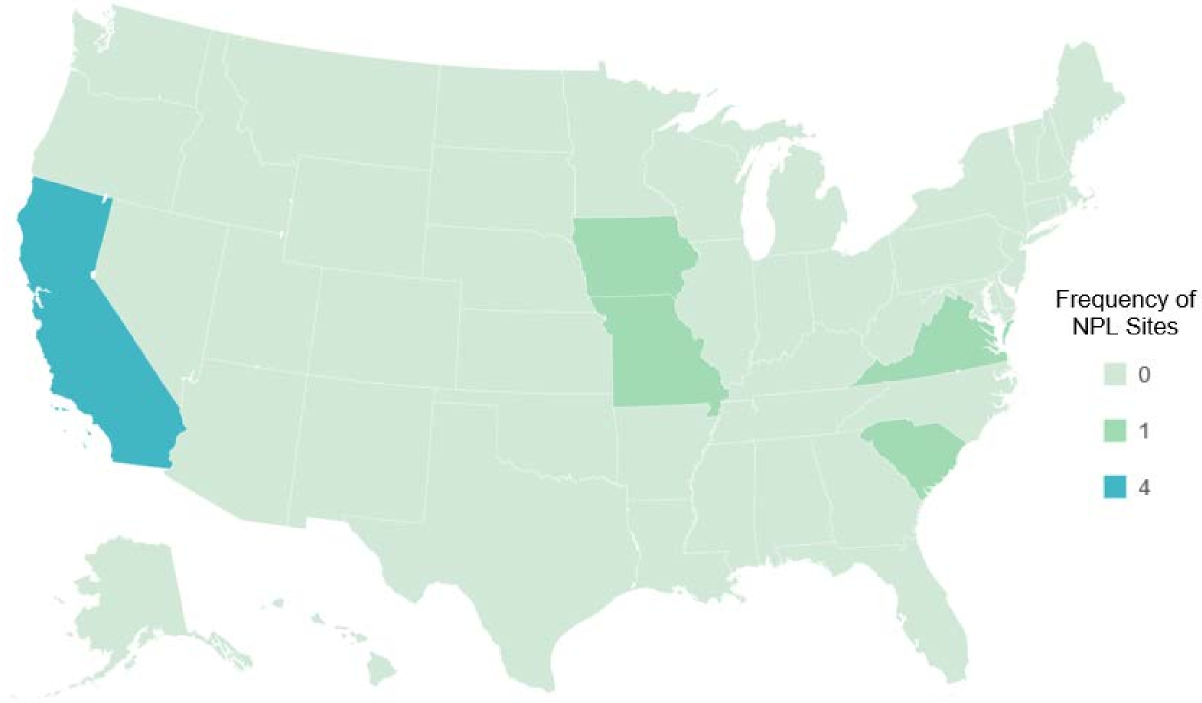
CHAPTER 5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

Disulfoton has been identified in at least 8 of the 1,867 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2019). However, the number of sites evaluated for disulfoton is not known. The number of sites in each state is shown in Figure 5-1.

Figure 5-1. Number of NPL Sites with Disulfoton Contamination



Source: ATSDR 2019

- Disulfoton was cancelled for use as a pesticide in the United States in 2009 by the EPA, and remaining stocks were permitted to be sold until 2011. Reported use of products containing disulfoton continued through 2016.
- The potential for human exposure to disulfoton is expected to be low for the general population.
- Disulfoton is a systemic organophosphate insecticide/acaricide used for agricultural purposes.
- Despite having a short predicted half-life in air (~3 hours), disulfoton may be transported long distances in the atmosphere; it has been detected in regions up to hundreds of kilometers away from countries where it is still is use as a pesticide.

- The detection of disulfoton in soil and the mobility of its degradation products suggest the potential for disulfoton to leach into groundwater.
- People who live near hazardous waste sites containing disulfoton may be at a higher risk of exposure than the general population, as are people who manufacture or handle disulfoton.

Disulfoton is not permitted for use as a pesticide in the United States as of 2009 following voluntary cancellation orders from a few companies that produce disulfoton-containing pesticides. Remaining stocks were permitted to be sold until 2011. Agricultural use of disulfoton-containing pesticides in the United States in 2016 was estimated by the U.S. Geological Survey in Virginia and North Carolina; no use of disulfoton-containing pesticides has been reported since 2016 (USGS 2021). Additionally, its use abroad may still continue. Historically, workers in industries that manufactured and formulated disulfoton, farm workers who entered treated fields after the insecticide was applied, and applicators of the insecticide were at a higher risk of exposure than the general population. However, this risk is no longer a concern in the United States given that production of disulfoton is no longer permitted and its current use is not likely. Due to the potential presence of disulfoton at hazardous waste sites, exposure may be possible for populations that live near these sites.

Disulfoton entered the environment primarily during its use as an insecticide/acaricide in crops and vegetables, and in home gardens. Other important pathways for disulfoton entry into the environment were the disposal of liquid disulfoton wastes into soil evaporation pits, ditches, ponds (Winterlin et al. 1989), and hazardous waste sites. Considering entry pathways and chemical and biological properties of disulfoton, soil is the environmental medium most likely to have been contaminated with disulfoton. The processes that may transport disulfoton from soil to other environmental media include leaching to groundwater, runoff to surface water, and absorption by plants (Holden 1986; Mostaghimi et al. 1993; Nash 1974; Plumb 1991; Sanborn et al. 1977; Spalding and Snow 1989). Biodegradation, abiotic hydrolysis and, to a lesser extent, sensitized oxidation are principally responsible for the loss of disulfoton from water (Capel et al. 1988; Mossman et al. 1988; Wanner et al. 1989). In a chemical spill in the Rhine River where an initial disulfoton concentration of 5 μ/L was observed, the estimated biodegradation half-life of disulfoton in 10ºC river water was 7–41 days (Wanner et al. 1989). The measured whole-body bioconcentration factor (BCF) for disulfoton in carp was 450, but disulfoton residues disappeared rapidly from the fish when they were placed in uncontaminated water (Takase and Oyama 1985). Biodegradation and photosensitized oxidation play major roles in the loss of disulfoton from soil (Gohre and Miller 1986; Wanner et al. 1989; Zepp et al. 1981). The estimated half-life of disulfoton in soil ranges from 3.5 to

≤290 days (Chapman et al. 1993, 1994a; Garg and Sethi 1980; Greenhalgh 1978; Harris et al. 1988; Jury et al. 1987a; Menzie 1972).

In the past, when it was still in use, disulfoton was detected at a maximum of 4.7 ng/m^3 in 1 of 123 ambient air samples from 10 locations in the United States (Carey and Kutz 1985). Disulfoton was qualitatively detected in groundwater samples from 1 of 479 hazardous waste sites (Plumb 1991) and in runoff water in an agricultural watershed at concentrations ranging from trace to 0.4 μg/L (Spalding and Snow 1989). Disulfoton was also detected in groundwater samples from the Nomini Creek Watershed in Virginia at a mean and maximum concentration of 0.39 and 2.87 μg/L, respectively (Mostaghimi et al. 1993). A core soil sample taken from a waste evaporation pit at a depth of 90 cm contained disulfoton at a concentration of 44 mg/kg (Winterlin et al. 1989). The mean concentration of disulfoton in the bottom soil of an agricultural tail water pit used to collect irrigation runoff was 13.4 μg/kg.

According to the Pesticide Residue Monitoring Program reports, disulfoton has not been detected in food in the United States in recent years (FDA 2017a, 2017b, 2018, 2019, 2022). The USGS estimated use of disulfoton on agriculture in the United States including vegetables and fruit up until 2016; no use has been reported since 2016 (USGS 2021). In the past, the Food and Drug Administration (FDA) estimated the average dietary intake of disulfoton for 1986–1991 for a 14- to 16-year-old male in the United States at 0.2 ng/kg body weight/day, a quantity over 1,000 times lower than the Food and Agricultural Organization of the United Nations/World Health Organization’s (FAO/WHO) acceptable daily intake (ADI) of 300 ng/kg body weight/day (EPA 1993; FAO/WHO 1991; Winter 1992; Yess 1991).

5.2 PRODUCTION, IMPORT/EXPORT, USE, AND DISPOSAL

5.2.1 Production

Disulfoton production in the United States is expected to have ceased due to its cancellation for use as a pesticide by EPA in 2009. Disulfoton can be produced commercially by a reaction of the sodium salt of O,O’-diethylhydrogen phosphorodithioate with 2-chloroethylthioethyl ether (VonRumker et al. 1974). No information is available in the Toxics Release Inventory (TRI) database on facilities that manufacture or process disulfoton because this chemical is not required to be reported under Section 313 of the Emergency Planning and Community Right-to-Know Act (Title III of the Superfund Amendments and Reauthorization Act of 1986) (EPA 2005). Information on current production volume is also not available in EPA’s Chemical Data Reporting database (CDR 2016). Following its cancellation by the

EPA in 2009, remaining stock was allowed to be sold and distributed by registrants until 2011 (EPA 2010). Disulfoton is listed as available for purchase through various chemical vendors, likely for laboratory use; however, there is insufficient information to determine where it is being produced (NLM 2021). According to the National Pesticide Information Retrieval System, there are currently no products containing disulfoton manufactured, sold, or used in the United States (NPIRS 2021).

5.2.2 Import/Export

Import and export data for disulfoton in recent years were not located, likely because disulfoton use was cancelled by EPA in 2009 (EPA 2010). An analysis of shipping records from 2001 to 2003 indicated that U.S. exports of disulfoton in that time period ranged from 118,573 to 288,054 pounds (Smith et al. 2008).

5.2.3 Use

Disulfoton is a systemic organophosphate insecticide/acaricide (i.e., it is absorbed and translocated by treated plants) effective for controlling a variety of harmful insects that attack many field and vegetable crops. Use of disulfoton products in the United States has likely ceased since its 2009 cancellation by EPA; however, EPA registrants were allowed to sell and distribute remaining stock until 2011, and non-registrants were allowed to use, sell, and distribute disulfoton products until they ran out (EPA 2010). As emulsifiable concentrates and in granular or pelleted/tableted forms, disulfoton was previously used to treat seeds and was applied to soils or plants. Disulfoton was also available in a ready-to-use liquid formulation (EPA 1984b). Historically, disulfoton was used to protect small grains, sugar cane, sorghum, Brazilian coffee crops, corn, cotton, cole, root, seed, forage, and other field crops; some vegetable, fruit (strawberry, pineapple), and nut crops; and forest plantings, ornamental, and potted plants (de Faria et al. 2016; EPA 1984b; VonRumker et al. 1974). Agricultural uses accounted for most of its consumption; small quantities were used on home and garden plants and for other purposes, such as mosquito abatement (VonRumker et al. 1974; Warnick and Eldredge 1972).

5.2.4 Disposal

The two preferable methods for disposing of wastes containing disulfoton are incineration and alkaline hydrolysis (NLM 2021). For disposal of low-viscosity wastes (permitting atomization in the combustion chamber) containing disulfoton, liquid injection incineration at 650–1,600°C and a residence time of 0.1–2 seconds are recommended. For the disposal of viscous and solid wastes, rotary kiln incineration at 820–

1,600°C and a residence time of seconds to hours, or fluidized bed incineration at 450–980°C with a residence time of seconds or longer are recommended. The effluent gases from the incineration units should pass through scrubbers or other air pollution control devices (NLM 2021). Alkaline hydrolysis leads to the complete degradation of disulfoton to non-toxic end products (alkaline salts of O,O-dimethylphosphorothioic acid and ethylthioethyl mercaptan). Acid hydrolysis produces essentially the same end products; however, the reaction rate is much slower (IRPTC 1985; Sittig 1980). Fifty percent hydrolysis at 70°C requires 60 hours at pH 5, but only 7.2 hours at pH 9 (Sittig 1980). In the alkaline hydrolysis method, the waste should be subjected to hydrolysis with 6% potassium hydroxide in isopropanol under reflux for 30 minutes (IRPTC 1985) or 5% sodium hydroxide in ethanol for 3 hours (for 2, 10, and 50% granular formulations) (Dillon 1981). The hydrolyzed product should be adsorbed on vermiculite, then incinerated or disposed of in a landfill (IRPTC 1985).

The EPA proposed incineration as the best demonstrated available technology (BDAT) for treating organophosphorus non-wastewaters (waste containing >1% by weight total suspended solids and >1% by weight total organic carbon). EPA demonstrated that rotatory kiln incineration at 1,000°C was satisfactory for attaining the proposed treatment standard of a maximum 0.1 mg/kg disulfoton in treated non-wastewaters (EPA 1989).

5.3 RELEASES TO THE ENVIRONMENT

The Toxics Release Inventory (TRI) data should be used with caution because only certain types of facilities are required to report (EPA 2005). This is not an exhaustive list. Manufacturing and processing facilities are required to report information to the TRI only if they employ ≥10 full-time employees; if their facility is included in Standard Industrial Classification (SIC) Codes 10 (except 1011, 1081, and 1094), 12 (except 1241), 20–39, 4911 (limited to facilities that combust coal and/or oil for the purpose of generating electricity for distribution in commerce), 4931 (limited to facilities that combust coal and/or oil for the purpose of generating electricity for distribution in commerce), 4939 (limited to facilities that combust coal and/or oil for the purpose of generating electricity for distribution in commerce), 4953 (limited to facilities regulated under RCRA Subtitle C, 42 U.S.C. section 6921 et seq.), 5169, 5171, and 7389 (limited S.C. section 6921 et seq.), 5169, 5171, and 7389 (limited to facilities primarily engaged in solvents recovery services on a contract or fee basis); and if their facility produces, imports, or processes ≥25,000 pounds of any TRI chemical or otherwise uses >10,000 pounds of a TRI chemical in a calendar year (EPA 2005).

5.3.1 Air

There is no information on releases of disulfoton to the atmosphere from manufacturing and processing facilities because these releases are not required to be reported (EPA 2005).

Disulfoton was cancelled for use in 2009, and it is unlikely to be released to the air in the United States (EPA 2010). Previously, disulfoton entered the atmosphere during its production and application as an insecticide (CPCR 1992).

5.3.2 Water

There is no information on releases of disulfoton to water from manufacturing and processing facilities because these releases are not required to be reported (EPA 2005).

Disulfoton was cancelled in 2009, and it is unlikely to be currently released to surface water or groundwater in the United States (EPA 2010). In the past, potential sources of release into surface water include wastewater discharge and runoff presumably from facilities involved in disulfoton manufacturing, formulation, and packaging (EPA 2008). Also, when disulfoton was still in use, leaching and runoff from treated fields and pesticide disposal pits had the potential to contaminate groundwater and surface water with disulfoton.

5.3.3 Soil

There is no information on releases of disulfoton to soil from manufacturing and processing facilities because these releases are not required to be reported (EPA 2005).

Disulfoton was cancelled in 2009, and it is unlikely to be released to soil in the United States (EPA 2010). Previously, disulfoton was released to agricultural, home, and garden soil during direct soil or foliar treatment with the insecticide and from disposal of disulfoton-containing wastes in hazardous waste sites (Kadoum and Mock 1978).

5.4 ENVIRONMENTAL FATE

5.4.1 Transport and Partitioning

Air. There is a paucity of experimental data regarding the transport and partitioning of disulfoton in air. Given the vapor pressure of 9.75×10^{-5} mm Hg at 25°C (NLM 2021), disulfoton should exist almost entirely in the vapor phase in the atmosphere (Eisenreich et al. 1981). Due to this, as well as its low particle diameter, the removal rate by dry deposition is expected to be low (Schroeder et al. 1987). Therefore, despite a short predicted half-life of 3 hours in the atmosphere (Atkinson 1988), vapor-phase disulfoton may travel long distances in the air depending on its reactivity characteristics. The detection of disulfoton in regions where it is not used suggests that its presence is due to atmospheric transport and deposition (Asman et al. 2005; Muir et al. 2004). Muir et al. (2004) estimated an empirical half-distance of 949 km for disulfoton. Asman et al. (2005) concluded that disulfoton may have been transported at least 500 km in order to be detected in rainwater in Denmark, where it is no longer sold. Muir et al. (2004) also calculated characteristic travel distance (CTD) and spatial range in air (SR_{air}) as values of indicators of long-range transport potential (LRTP) using three model scenarios: default conditions, [OH] reduced 10-fold, and intermittent precipitation. Under default conditions according to the model, the CTD of disulfoton is 20–21 km, and the SR_{air} is 2% of the earth's circumference. When atmospheric degradation rates are lowered by a factor of 10, CTD increases to 188–199 km, and SR_{air} increases to 7%. When accounting for intermittent precipitation, CTD increases to 193–206 km, and SR_{air} remains at 7% (Muir et al. 2004). Hayward et al. (2010) studied concentrations of disulfoton in Egbert, Ontario and calculated the CDT of disulfoton as 207 km. Muir et al. (2004) notes that the uncertainty in atmospheric degradation rates is large enough to account for the discrepancy in estimated and observed long-range transport of the pesticide.

The solubility of 25 mg/L (NLM 2021) ensures that at least partial removal of atmospheric disulfoton will occur by wet deposition. Disulfoton has been detected in precipitation (Asman et al. 2005; Kurt-Karakus et al. 2011).

Water. The transport of disulfoton from water to air can occur due to volatilization. Volatilization from water is slow to negligible for compounds with a Henry's law constant of $<10^{-5}$ atm-m^3/mol (Thomas 1990). Therefore, disulfoton, with a Henry's law constant value of 2.2×10^{-6} atm-m^3/mol (NLM 2021), will volatilize slowly from water. The rate of volatilization increases as the water temperature and ambient air flow rate (wind) increases and decreases as the rate of adsorption on sediment and suspended solids increases (Dragan and Carpov 1987). The estimated gas-exchange half-life for disulfoton volatilization from the Rhine River at an average depth of 5 m at 11°C was 900 days (Wanner et al. 1989). The estimated volatilization half-life of an aqueous suspension of microcapsules containing disulfoton at 20°C with still air was >90 days (Dragan and Carpov 1987).

Sediment and Soil. Adsorption to particulate matter will transport disulfoton from water to suspended solids and sediment in water. The organic carbon-adjusted soil sorption coefficient (K_{oc}) for disulfoton varies between 600 and 2,612 (Gawlik et al. 2000; Gramatica et al. 2000; Wauchope et al. 1992). This range of K_{oc} values suggests that disulfoton in water adsorbs moderately to strongly to suspended solids and sediments (Swann et al. 1983), and this process may facilitate transport of disulfoton.

The transport processes that may move disulfoton from soil to other media are volatilization, leaching, runoff, and absorption by plants. Volatilization of disulfoton from wet soil may be greater than from relatively dry soil (Gohre and Miller 1986). Like other pesticides, disulfoton in soil partitions between soil-sorbed and soil-water phases (Racke 1992). This latter phase may be responsible for the volatilization of disulfoton from soil; however, due to the low Henry's law constant value, the rate of disulfoton volatilization from the soil-water phase to the atmosphere would be low.

The reported K_{oc} values of 600–2,612 suggest that the adsorption of disulfoton to soil is moderate to strong and that the rate of leaching may be minor in most soils. Batch-type adsorption tests and soil column studies showed that the disulfoton adsorption rate in soil increases as the clay content of the soil increases (King and McCarty 1968; McCarty and King 1966). Disulfoton leaching through Hugo sandy loam soil was initially rapid, but very little further leaching was observed with an increase in eluent volume (McCarty and King 1966). For example, 27.5% of disulfoton applied to a 6-inch soil column eluted with a total of 4 feet of buffered water (pH 7), but only 29% eluted with a total of 110 feet of buffered water. Other investigators concluded from soil column and soil thin-layer chromatography studies that disulfoton is only very slightly to moderately mobile in soil (de Faria et al. 2016; Harris 1969; Helling et al. 1974; Thornton et al. 1976). Mobility may decrease with an increase in soil pH and organic content (Thornton et al. 1976). The oxidation products of disulfoton (sulfone and sulfoxide) are less mobile in soils than the parent compound (EPA 1989). Due to increased polarity, the mobility of the oxidation products is expected to depend on the soil's cation exchange properties; mobility would decrease as the soil's cation exchange potential increases. Disulfoton has been detected infrequently and at low concentrations in groundwater from agricultural soil (Holden 1986; Mostaghimi et al. 1993) and in groundwater from disposal sites (Plumb 1991). These observations suggest that small amounts of disulfoton leach through certain soils into groundwater. Disulfoton sulfone and an oxygen analogue of disulfoton sulfone, degradation products of disulfoton with comparable toxicity and higher solubility, were observed moving into deeper soil layers, suggesting that disulfoton and its potential metabolites can reach groundwater even if the sorption is significant or even dominant (de Faria et al. 2016).

Disulfoton is also transported through soils or from soil to surface water (streams or rivers) via runoff. Pesticides with water solubilities >10 mg/L move mainly in solution phase in runoff water (Racke 1992). Disulfoton, with a water solubility of 25 mg/L (NLM 2021), is expected to be found mainly in runoff water. In a runoff event from agricultural soil in Nebraska, low levels of disulfoton were detected both in the dissolved state and in eroding soil particles in the sorbed state (Spalding and Snow 1989).

Disulfoton is absorbed from soil by the root systems of plants and is translocated to the plant top (Nash 1974). Plants metabolize disulfoton to its sulfone, sulfoxide, and oxons (Szeto et al. 1983a, 1983b). The concentrations of disulfoton and its metabolites in plant tops depend on the applied dosage in soil and the type of plants. The level of parent compound and its metabolites reaches a maximum concentration in plants within days or weeks and then tends to decrease (Nash 1974; Szeto et al. 1983a,1983b). When disulfoton was applied to a soil at levels of 0.5 and 4.0 kg active ingredient per hectare in asparagus field plots, the levels of sulfone, sulfoxide, and oxons in asparagus ferns increased steadily to maximums of 14 and 61 mg/kg (fresh weight) in 70–85 days and then declined to 0.4 and 17.1 mg/kg in 147 days; no parent compound was detected at any time after 14 days following application (Szeto et al. 1983a). Similarly, the metabolites of disulfoton were detected in lettuce grown in a treated field (Szeto et al. 1983b). The residual levels of disulfoton and its metabolites in vegetables grown on treated soil were highest in carrots, intermediate in Chinese cabbage, and lowest in turnips (Sanborn et al. 1977). Chapman et al. (1994b) studied the effects of multiple soil applications of disulfoton (one treatment each year for 3 years) on enhanced microbial degradation in soil and subsequent uptake by seed potatoes and foliage. Disulfoton was the major insecticidal component detected in soil, a minor component of seed potatoes, and was not detected (<0.02 ppm) in potato foliage during all three treatment years. Disulfoton sulfoxide and sulfone were the major insecticidal degradation products detected in the seed potatoes and foliage.

Other Media. The bioconcentration of disulfoton and its oxidation products (sulfoxide and sulfone) in carp (*Cyprinus carpio*) was investigated in a continuous flow water system for ≤56 days (Takase and Oyama 1985). The whole-body BCF values in carp were ≈450 for disulfoton, <1 for the sulfoxide, and <6 for the sulfone. Disulfoton disappeared rapidly from fish tissues when the fish were transferred to uncontaminated fresh water. BCFs of disulfoton in fish based on concentrations at 168 hours were 261 (female guppy), 482 (male guppy), 202 (killifish), 96.7 (goldfish), and 127 (white cloud mountain fish) (Tsuda et al. 1997). Since estimated BCF values for disulfoton are below 1,000, disulfoton is not considered to be bioaccumulative. A microcosm, simulating paddy fields containing water, sweet potato, tobacco cutworm (*Spodoptera litura*), algae (*Spirogyra crassa*), red snail (*Indoplanorbis exustus*), Daphnia, mosquito larvae (*Culex pipiens*), and guppies (*Labistes reticulatus*), was used to assess disulfoton accumulation in aquatic organisms over a 33-day period (Tomizawa 1980). Whole-body BCF values of 9 and 2,487 were reported for snails and guppies, respectively.

5.4.2 Transformation and Degradation

Air. One of the important reactions for most organic pollutants in the atmosphere is with hydroxyl radicals. Using an estimation method (Meylan and Howard 1993), the estimated rate constant for the vapor-phase reaction of disulfoton with hydroxyl radicals is 13.2×10^{-11} cm^3/molecule-second. At an average atmospheric hydroxyl radical concentration of 5×10^5 radicals/cm^3 (Atkinson 1988), the estimated half-life of disulfoton in the atmosphere due to this reaction is about 3 hours. Disulfoton is not susceptible to direct photolysis in sunlight (Gohre and Miller 1986). As with soil and water (Gohre and Miller 1986; Zepp et al. 1981), it is possible that disulfoton reacts with singlet oxygen in the atmosphere.

Water. The three processes responsible for the transformation and degradation of disulfoton in water are abiotic hydrolysis, photosensitized oxidation, and biodegradation. Estimated hydrolysis half-lives were 103 days at 25°C and pH 7 (EPA 1988) and 170 days at 11°C and pH 7.9 (Wanner et al. 1989). Hydrolysis products of disulfoton are diethylthiophosphoric acid and 2-ethylmercaptothio ether (Muhlmann and Schrader 1957).

Direct photolysis of disulfoton is negligible since it does not significantly absorb sunlight (Wanner et al. 1989). Disulfoton is more likely to react with singlet molecular oxygen ($^1O_2$) produced from the reaction of certain photochemically excited dissolved organic matter (e.g., humic and fulvic substances) with molecular oxygen in water (Zepp et al. 1981). The estimated near-surface half-life for photosensitized oxidation of disulfoton by sunlight available during midwinter in the southern United States was 3 hours (Zepp and Baughman 1978). Due to light attenuation with increasing water depth, the half-life of disulfoton due to the oxidation reaction is expected to increase with increasing water depth. Estimated half-lives due to photosensitized oxidation are 1,000 days at a depth of 5 m for a winter day and 100 days for a summer day (Wanner et al. 1989). The products of photosensitized oxidation are disulfoton sulfone and disulfoton sulfoxide (Mitchell et al. 1968). Hydroxyl radicals in natural water also oxidize disulfoton. When a 13 µm solution of disulfoton was exposed to October sunlight (Davis, California) in the presence of 100 µm hydrogen peroxide, 49% of the insecticide disappeared in 10.2 days due to reaction with hydroxyl radicals (Draper and Crosby 1984). Since eutrophic water samples of the same type studied generate hydrogen peroxide levels 30 µM or lower, the rate of this reaction will be slower in natural surface water (Draper and Crosby 1984).

Following an accidental discharge of stored chemicals including disulfoton, the estimated biodegradation half-life of disulfoton in Rhine River water was between 7 and 41 days at 10°C (Wanner et al. 1989). Therefore, biodegradation of disulfoton in water is expected to be important, and the rate will depend on the initial concentration. A theoretical model predicted that over 12 days biodegradation and photolysis would account for an 80% mass loss of disulfoton in the Rhine River after an accidental spill incident (Mossman et al. 1988); however, the removal of disulfoton by chemical processes was much slower than by biodegradation (Capel et al. 1988).

Sediment and Soil. Disulfoton in soil and sediment may undergo degradation and transformation by hydrolysis, photoinduced oxidation, and biotic processes. The hydrolysis of disulfoton may occur in the soil/sediment-water phase, as opposed to the soil/sediment-sorbed phase. As a result, the rate of hydrolysis is expected to be comparable to that in water. Based on slow hydrolysis rates observed in water, hydrolysis of disulfoton in soil is not expected to be significant. A group of investigators reported the oxidation of disulfoton on soil surfaces by singlet oxygen produced from sunlight irradiation (Gohre and Miller 1986; Hebert and Miller 1990; Miller et al. 1989). The initial loss of disulfoton on soil surfaces by photooxidation is quite rapid and slows down as the reaction proceeds. Thus, attributing the loss to a first-order rate process and assigning a half-life to this process is misleading (Miller et al. 1989). Although the most rapid oxidation occurred in soil with the lowest organic carbon, half of the original concentrations of disulfoton in four different soil samples was lost in ≈3 days (Gohre and Miller 1986). The rate substantially decreased over the course of irradiation. The photooxidation of disulfoton occurred appreciably deeper than optical depths (depths for sunlight penetration in soil) of 0.2–0.3 mm (Hebert and Miller 1990). In aerated and moisture-unsaturated soil, the photooxidation can proceed up to a soil depth of 2 mm (Hebert and Miller 1990). The primary photooxidation product was the sulfoxide with trace amounts of the sulfone (Gohre and Miller 1986).

In laboratory tests, several fungi and cultures of actinomycetes isolated from garden soil readily degraded disulfoton (Bhaskahan et al. 1973). In flooded soil under anaerobic conditions, the reduction of disulfoton to disulfoton sulfoxide or disulfoton sulfide was due to biological conversion (Tomizawa 1975). Since the bacterial populations in sediments and soils are higher than in typical surface waters (Mossman et al. 1988), biodegradation is expected to play a major role in the loss of disulfoton in soil and sediment, as occurred in the disulfoton spill in the Rhine River (Capel et al. 1988; Wanner et al. 1989).

Several investigators have reported the rate of overall loss of disulfoton from soil due to all biotic and abiotic processes. The estimated half-life of disulfoton in soil ranged from 3.5 to 14 days (Chapman et al. 1993; Garg and Sethi 1980; Greenhalgh 1978; Harris et al. 1988; Jury et al. 1987a; McCarty and King 1966; Rao et al. 1985; Shaw 1975), although half-life values of 17 and 42 days were reported for loam and Plainfield sand, respectively (Chapman et al. 1994a). A half-life value of ≤290 days was also reported for soil (soil type unspecified) (Menzie 1972). The estimated persistence of disulfoton, defined as the concentration of disulfoton remaining elevated or constant in soil, varied between 28 and >64 days (Belanger and Hamilton 1979; Clapp et al. 1976; Jury et al. 1987b; Kearney et al. 1969). Soil type and soil temperature influenced the degradation rate of disulfoton. Disulfoton degraded almost twice as fast over the first 12 weeks post-application in loam as compared to Plainfield sand; however, the authors believe that lower temperatures may have contributed to the slower disappearance of disulfoton in the Plainfield sand study (Chapman et al. 1994a). Since the compound degraded faster during winter in Evesboro loamy sand soil than during summer in Chillum silt loam soil, the authors (Menzer et al. 1970) concluded that soil type was predominantly responsible, rather than temperature.

The presence of light and higher soil pH (pH 8 versus 5) also accelerated degradation of disulfoton in soil (Shaw 1975). The metabolites isolated from disulfoton degradation in soil were the sulfoxide and sulfone (Chapman et al. 1994a, 1994b); minute amounts of oxons were found (Clapp et al. 1976; Greenhalgh 1978; Shaw 1975; Szeto et al. 1983a). Diethyl phosphorothioate was identified as the major metabolite in the aqueous fraction of soil (Shaw 1975). Disulfoton and disulfoton sulfoxide degraded in ≤32 days in soil, while disulfoton sulfone persisted for >64 days (Clapp et al. 1976; Greenhalgh 1978).

5.5 LEVELS IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to disulfoton depends, in part, on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of disulfoton in unpolluted atmospheres and in pristine surface waters are often so low as to be near the limits of current analytical methods. In reviewing data on disulfoton levels monitored or estimated in the environment, it should also be noted that the amount of chemical identified analytically is not necessarily equivalent to the amount that is bioavailable.

Table 5-1 shows the lowest limit of detections that are achieved by analytical analysis in environmental media. An overview summary of the range of concentrations detected in environmental media is presented in Table 5-2.

Table 5-1. Lowest Limit of Detection for Disulfoton Based on Standards[a]

Media	Detection limit	Reference
Environmental media		
Wastewater	5.9 ng/L	Basheer et al. 2007
Surface water	0.0006 ng/L	Kurt-Karakus et al. 2011
Drinking water	0.3 µg/L	Edgell et al. 1991
Groundwater	1.9 ng/L	WQP 2021
Ambient air	5.7 pg/m^3	Kurt-Karakus et al. 2011
Sediment	≤0.1 mg/kg	Belisle and Swineford 1988
Food products and crops		
Soil, asparagus tissue	0.01 mg/kg	Szeto and Brown 1982
Cereal, maize, and wheat	0.55 µg/kg	Gonzalez-Curbelo et al. 2017
Strawberries	0.05 µg/g	Baldim et al. 2012
Rice, wheat, buckwheat, and dried beans	0.3 µg/kg	Aoki et al. 1975
Cow milk (disulfoton and five metabolites as total residue)	1 µg/kg	Bowman and Beroza 1969
Beverages	0.10 µg/L	dos Anjos and de Andrade 2014
Human serum		
Whole blood	0.90 ng/mL	Usui et al. 2012
Blood	0.01 µg/g	Musshoff et al. 2002
Urine	0.46 ng/mL	Usui et al. 2012

Table 5-2. Summary of Environmental Levels of Disulfoton

Media	Low	High	For more information
Outdoor air (pg/m^3)	<5.7	67.4	Table 5-3 or Section 5.5.1
Surface water (ng/L)	<2	3,300	Table 5-4 or Section 5.5.2
Ground water (ng/L)		<60	Section 5.5.2
Rain water (ng/L)	<1.2	<60	Section 5.5.2

Detections of disulfoton in air, water, and soil at NPL sites are summarized in Table 5-3.

Table 5-3. Disulfoton Levels in Water, Soil, and Air of National Priorities List (NPL) Sites

Medium	Median[a]	Geometric mean[a]	Geometric standard deviation[a]	Number of quantitative measures	NPL sites
Water (ppb)	NA	NA	NA	NA	NA
Soil (ppb)	430,000	383,000	17.5	7	4
Air (ppbv)	NA	NA	NA	NA	NA

[a] Concentrations found in ATSDR site documents from 1981 to 2019 for 1,867 NPL sites (ATSDR 2019). Maximum concentrations were abstracted for types of environmental media for which exposure is likely. Pathways do not necessarily involve exposure or levels of concern.

5.5.1 Air

No data were located measuring disulfoton in air after its cancellation in 2009 as the concentration in the United States is now expected to be lower compared to data from prior to 2009. In Egbert, Ontario where disulfoton was in use during the time of the study, disulfoton was detected in air at a mean concentration of 50.7 pg/m^3 (Hayward et al. 2010). Disulfoton concentrations in the air fell near or below quantitation limits during the winter, but were measured at much higher levels during the growing seasons between May and September (see Table 5-4) (Hayward et al. 2010). Disulfoton was not detected (detection limit 5.7 pg/m^3) in 11 passive air samples collected in 2004–2005 from five locations in the Great Lakes Region of Canada (Kurt-Karakus et al. 2011). Disulfoton was also not detected in 151 atmospheric samples collected April to September 1995 at seven locations across the Midwestern United States (Foreman et al. 2000). Disulfoton was detected at maximum and mean concentrations of 4.7 and 0.1 ng/m^3 in only 1 of 123 ambient air samples collected from 10 locations in the United States in 1980 (Carey and Kutz 1985; Kutz 1983; Kutz and Carey 1986). Measured concentrations of disulfoton in air are presented in Table 5-4.

Table 5-4. Outdoor Air Monitoring Data for Disulfoton

Location	Geographic type	Date(s)	Range	Mean concentration	Notes	Reference
Egbert, Ontario	Rural agricultural	March 2006–September 2007	27.5–67.4 pg/m^3	50.7 pg/m^3	Range and mean (arithmetic means of both active sampling techniques) is during growing periods (May–September)	Hayward et al. 2010
Ontario	Not specified	2004–2005	<5.7 pg/m^3			Kurt-Karakus et al. 2011
Mississippi, Iowa, Minnesota, Michigan	Urban and agricultural	1995 April–September	Not detected		Weekly samples (151 total); detection limit not reported	Foreman et al. 2000
10 U.S. locations[a]	Not specified	1980	≤4.7 ng/m^3	0.1 ng/m^3	1 out of 123 samples (0.8%) were positive for disulfoton	Carey and Kutz 1985; Kutz 1983; Kutz and Carey 1986

[a]Columbia, South Carolina; Lubbock, Houston, and Harlingen, Texas; Huntsville, Alabama; Pasadena and Fresno, California; Mississippi State, Mississippi; Helena, Montana; and Pekin, Illinois.

5.5.2 Water

Disulfoton was not reported above the lower quantification limit of 1.9–60 ng/L (ppt) in over 1,100 ambient surface water data points compiled for 2020–2021 from EPA STOrage and RETrieval (STORET) and National Water Information System (NWIS databases (WQP 2021). In a study of the Yakima River Basin, Washington, May 1999 through January 2000, disulfoton was detected in 4 of 98 river water samples (USGS 2002). The minimum concentration was <17 ng/L and the maximum concentration was estimated to be 3,300 ng/L. Estimated usage of disulfoton in Granger Drainage Basin, part of the Yakima River Basin, for 1999 was 6,100 pounds applied to asparagus (USGS 2002). Between 1998 and 2001, 30 lakes in Canada and the northeastern United States were sampled for current-use pesticides, including disulfoton, using a method with a much lower detection limit of 0.002 ng/L (Muir et al. 2004). Disulfoton was detected in lakes with agricultural inputs as well as remote lakes, with no distinct differences observed between agricultural input lakes and other lakes at mid-latitude. Disulfoton was detected in 40 of 164 surface water samples collected in 2003–2005 from 10 inland lakes located in Ontario, Canada at concentrations of <0.006–1.8 ng/L (Kurt-Karakus et al. 2011).

Disulfoton was not reported at or above the lower quantification limit of 60 ng/L (ppt) in over 600 groundwater data points compiled for 2020–2021 from EPA STORET and NWIS databases (WQP 2021). In a pesticide analysis study conducted from 2016 to 2018 of 54 shallow monitoring wells located in Nassau/Queens and Suffolk counties, Long Island, disulfoton was detected in 1 well (Fisher et al 2021). This well was located in a pesticide management area with mixed agriculture and horticulture crops; the concentration was not reported. In the past, disulfoton was detected in 7 groundwater samples from 28 California counties at a maximum of 6 µg/L from May 1979 to April 1984 (Cohen 1986; Hallberg 1989; Holden 1986) and in a 1985 survey of groundwater in the Nomini Creek Watershed in Virginia (range 0.39–2.87 µg/L) (Mostaghimi et al. 1993).

Disulfoton was not reported at or above the lower quantification limit of 60 ng/L (ppt) in 25 rainwater samples collected in Minnesota in 2020 from EPA STORET and NWIS databases (WQP 2021). Disulfoton was not detected (detection limit not reported) in 32 precipitation samples collected April to September 1995 at seven locations across the Midwestern United States (Coupe et al. 2000). Disulfoton was detected in 10 of 51 rainwater samples collected in 2003–2005 from the Great Lakes Region of Ontario, Canada at concentrations of <0.0012–3.8 ng/L (Kurt-Karakus et al. 2011). Between January 2000 and July 2001, disulfoton was detected, but not quantified, in rainwater in Roskilde, Denmark and in rainwater at concentrations up to 7 ng/L in Oure, Denmark, despite it being banned and not sold in the country. The sum deposition was calculated to be 0 ng/m/year in Roskilde and 1,169 ng/m/year in Oure (Asman et al. 2005). Surface water monitoring data for disulfoton are presented in Table 5-5.

5.5.3 Sediment and Soil

Only NPL data measured disulfoton in sediment and soils after 2009 and is reported in Table 5-2. All other located data are of samples taken prior to the cancellation of disulfoton and levels are expected to now be lower in the United States. The primary method for the disposal of liquid pesticide wastes in California in the past has involved soil evaporation pits, ditches, and ponds (Winterlin et al. 1989). A core soil sample taken from one such pit in northern California contained 44 mg/kg disulfoton at a depth of 90 cm (Winterlin et al. 1989). In the Salton Sea, a manmade lake in California designated by the state as an agricultural drainage reservoir, the concentration of disulfoton in sediment was less than the detection limit of 0.20 ng/g dry weight at all locations sampled in 2000 (Sapozhnikova et al. 2004). Concentrations were measured again in 2001 and disulfoton was found at less than the detection limit (0.20 ng/g dry weight) in four locations sampled and at 29.6 ng/g dry weight in a fifth location sampled where the concentration was previously below the detection limit (Sapozhnikova et al. 2004).

Table 5-5. Surface Water Monitoring Data for Disulfoton

Location(s)	Geographic type	Date(s)	Range	Mean concentration	Notes	Reference
Turnbull, Wawanosh, Bells, Plastic, Opeongo, Wavy, Windy, Flack, Batchawana, Big Turkey Lakes	Ontario, Canada	2003–2005	<0.0006–1.8 ng/L	0.001	Disulfoton was detected in 24% of samples; detection limit: 0.0006 ng/L	Kurt-Karakus et al. 2011
Yakama River Basin: Moxee Drain; Granger Drain; Yakama River at Kiona	Washington	1999–2000	<17–3,300 ng/L	<17 ng/L	Disulfoton concentration at 90th percentile: <17 ng/L	USGS 2002
C1, A, Ward Hunt, A-A, D-J, BK-Z Lakes	Arctic lakes in Canada and Northeastern United States	1998–2001	<0.002–0.06 ng/L	0.01 ng/L	Disulfoton was detected in one out of six lakes; detection limit: 0.002 ng/L	Muir et al. 2004
Mista, Merrick, Shipiskan, Wuchuska, Big Trout, Fourmont, and Minipi Lakes	Sub-Arctic lakes in Canada and Northeastern United States	1998–2001	<0.002–3.0 ng/L	1.2 ng/L	Disulfoton was detected in five out of seven lakes; detection limit: 0.002 ng/L	Muir et al. 2004
Nipigon, Paguchi, Eva, Thunder, Sandybeach, Dasserat, St. Jean, Britt Brook, Virgin, Opeongo, and Cromwell Lakes; Connery Pond, Moose Pond, and Bates Pond	Remote mid-latitude lakes in Canada and Northeastern United States	1998–2001	<0.002–4.9 ng/L	0.76 ng/L	Disulfoton was detected in 10 out of 11 lakes; detection limit: 0.002 ng/L	Muir et al. 2004
Simcoe, Seneca, and Cayuga Lakes	Agricultural input lakes in Canada and Northeastern United States	1998–2001	<0.002–0.44 ng/L	0.23 ng/L	Disulfoton was detected in two out of three lakes; detection limit: 0.002 ng/L	Muir et al. 2004

Table 5-5. Surface Water Monitoring Data for Disulfoton

Location(s)	Geographic type	Date(s)	Range	Mean concentration	Notes	Reference
Grand River Basin, Saugeen River Basin, and Thames River Basin; Ontario, Canada	River surface water	January 1981–December 1985	Not detected	Not detected	Samples were collected year-round during storm runoff and base flow conditions; three to four samples were collected at the mouth of each river; detection limit: <0.1 µg/L	Frank and Logan 1988
Lake Huron (9 sites), North Channel (2 sites), Georgian Bay (5 sites), and Lake Superior (17 sites)	Upper Great Lakes surface water	Summer of 1974	Not detected	Not detected	Sampling stations were chosen based on proximity to major rivers, industrial plants, and municipal areas but were at least 1 km from shore as not to be immediately influenced by these source areas; 1 sample was taken at each station for a total of 33 samples; quantification limit: 0.003 µg/L	Glooschenko et al. 1976

Disulfoton was detected in sediment samples in Kafue Town (0.067 μg/g) and Kitwe (0.041 μg/g) in a study to determine the types of pesticides and herbicides in the Kafue River in Zambia (Syakalima et al. 2006). In 1974, disulfoton was detected at concentrations up to 227.8 ppb in nine bottom soil samples from tail water pits used to collect irrigation runoff from corn and sorghum fields in Haskell County, Kansas (Kadoum and Mock 1978). At a detection limit of 0.01 mg/kg, disulfoton was not detected in sediment samples collected from Lakes Superior and Huron, including Georgian Bay, in 1974 (Glooschenko et al. 1976). Disulfoton concentrations measured in soil and sediments are presented in Table 5-6.

Table 5-6. Concentrations of Disulfoton in Soil and Sediment

Location/date	Concentration	Reference
Salton Sea, California sediment		Sapozhnikova et al. 2004
May 2000		
Middle sampling location 1	<0.20 ng/g dry weight (<0.20 μg/kg)	
Southern sampling location 1	<0.20 ng/g dry weight (<0.20 μg/kg)	
Northern sampling location 1	<0.20 ng/g dry weight (<0.20 μg/kg)	
May 2001		
Middle sampling location 1	29.6 ng/g dry weight	
Southern sampling location 2	<0.20 ng/g dry weight (<0.20 μg/kg)	
Southern sampling location 3	<0.20 ng/g dry weight (<0.20 μg/kg)	
Northern sampling location 1	<0.20 ng/g dry weight (<0.20 μg/kg)	
Northern sampling location 2	<0.20 ng/g dry weight (<0.20 μg/kg)	
Haskell County, Kansas		Kadoum and Mock 1978
1974		
Pit bottom soil from samples serving corn fields		
Median	11.4 μg/kg	
Mean	13.8 μg/kg	
Maximum	32.7 μg/kg	
Bottom soil from samples serving corn and sorghum fields		
Mean	11.0 μg/kg	
Bottom soil from samples serving sorghum fields		
Median	117.2 μg/kg	
Mean	117.2 μg/kg	
Maximum	227.8 μg/kg	

5.5.4 Other Media

Under the National Lake Fish Tissue Study, the EPA estimated the concentration of selected persistent, bio accumulative, and toxic chemical residues in fish tissue from 500 sampling locations in the United States (EPA 2009b). Disulfoton was not detected in any of the samples (detection limit 161 μg/kg) from years 2000–2003. Disulfoton concentrations were measured in fish tissues from the Salton Sea, a manmade lake designated by the state as an agricultural drainage reservoir in California (Sapozhnikova et al. 2004). Mean disulfoton concentrations and standard deviations were 20±17 ng/g in liver, 17±16 ng/g in gonads, 7±8 ng/g in muscle, and 7±4 ng/g in gills. Disulfoton was observed in all tissues from Tilapia and Corvina in elevated concentrations compared to other studies, as noted by Sapozhnikova et al. (2004), and disulfoton was one of the most abundant pesticides observed. In Zambia, disulfoton was detected in fish muscle at mean concentrations of 0.020 μg/g in Chingola, 0.46 μg/g in Kitwe, and 0.034 μg/g in Kafue National Park (Syakalima et al. 2006). Results from these studies are further summarized in Table 5-7.

Table 5-7. Concentrations of Disulfoton in Fish Tissues

Location	Date(s)	Species/ tissue	Mean concentration	Range	Number of Samples	Reference
Salton Sea, California	May 2001	Corvina species				
		Muscle	6.7 ng/g wet weight	0.5–23.7 ng/g wet weight	6	
		Liver	19.7 ng/g wet weight	8.5–54.6 ng/g wet weight	6	
		Gonads	16.5 ng/g wet weight	2.3–46.9 ng/g wet weight	6	
		Gills	6.7 ng/g wet weight	3.5–15.3 ng/g wet weight	6	
Salton Sea, California	May 2001	Tilapia species				Sapozhnikova et al. 2004
		Muscle	7.8 ng/g wet weight	1.8–17.6 ng/g wet weight	9	
		Liver	31.0 ng/g wet weight	4.6–80.3 ng/g wet weight	9	
		Gonads	29.3 ng/g wet weight	5.0–52.3 ng/g wet weight	9	
		Gills	12.2 ng/g wet weight	0.9–34.2 ng/g wet weight	9	

Table 5-7. Concentrations of Disulfoton in Fish Tissues

Location	Date(s)	Species/ tissue	Mean concentration	Range	Number of Samples	Reference
Kafue River in Chingola, Zambia	2003–2005 (March–November of each year)	*Serranochromis angusticeps* Muscle	0.020 µg/g	Not reported	10	Syakalima et al. 2006
Kafue River in Kitwe, Zambia	2003–2005 (March–November of each year)	*S. angusticeps* Muscle	0.046 µg/g	Not reported	10	Syakalima et al. 2006
Kafue River in Kafue National Park, Zambia	2003–2005 (March–November of each year)	*S. angusticeps* Muscle	0.034 µg/g	Not reported	10	Syakalima et al. 2006
500 sampling locations in lakes and reservoirs in the U.S. lower 48 states	2000–2003	Predator composites Bottom-dweller composites	0 µg/kg 0 µg/kg	Not applicable Not applicable	486 395	EPA 2009b

In 2009, the EPA cancelled disulfoton as a pesticide, and it is no longer used on crops in the United States (EPA 2009b). However, use as a pesticide may continue abroad. The FDA's Pesticide Monitoring Program for domestic and imported foods reports that disulfoton residues have not been detected in recent years (FDA 2017a, 2017b, 2018, 2019, 2021b). Previously, the FDA's monitoring program for domestic and imported food commodities during fiscal years 1978–1982 detected disulfoton in unspecified foods at unspecified concentrations (Yess et al. 1991). During 1982–1986, the FDA Los Angeles District Laboratory detected disulfoton sulfone in 45 samples of 6,391 domestic agricultural commodities and in 1 sample of 12,044 imported agricultural commodities at concentrations ranging from 0.05–1.0 mg/kg (Luke et al. 1988). Disulfoton was not detected in various domestic food commodities by state regulatory monitoring activities during fiscal year 1988–1989 (Minyard and Roberts 1991). In a pesticide residue screening program conducted in 1989–1991 in San Antonio, Texas, on 6,970 produce samples, disulfoton was detected (0.1 ppm detection limit) in two produce samples (one sample of broccoli and one sample of cabbage) (Schattenberg and Hsu 1992). In a 1993 study of pesticide residue contamination of processed infant formula, disulfoton was not detected (detection limit <0.02 µg/g [ppm]) in 32 milk-based and 25 soy-based infant formulas (Gelardi and Mountford 1993).

Disulfoton has been detected in beverages. Disulfoton was detected below the limit of quantification (2.37 µg/L) in two brand names of coconut water, but was not detected in another brand name of coconut water or natural coconut tested by dos Anjos and de Andrade (2014). In a study of 15 white wine samples, disulfoton was detected in two samples at 3.53 and 5.78 µg/L; it was not found in any of four rose wine samples tested (dos Anjos and de Andrade 2015).

5.6 GENERAL POPULATION EXPOSURE

Since disulfoton was cancelled by the EPA in 2009 (EPA 2010), the general population in the United States is not likely to be exposed to disulfoton, although remaining stock was permitted to be sold until 2011 or until stock ran out, and agricultural use in the United States was estimated as recent as 2016 (USGS 2021). Its use abroad may continue, but it is unknown if uses abroad may lead to exposure of the general population as there is insufficient information to confirm its use. Disulfoton has been detected in soils at multiple hazardous waste sites in the United States, indicating populations living near these hazardous waste sites may be at risk of disulfoton exposure.

Disulfoton has been very infrequently detected in ambient air and at very low concentrations (see Section 5.5.1). Therefore, the exposure of the general population to disulfoton from inhaling ambient air is probably insignificant. Disulfoton has never been detected in drinking water (see Section 5.5.2). This is consistent with observations that it occurs at very low concentrations and has only infrequently been detected in groundwater (Fisher et al 2021; WQP 2021). Therefore, general population exposure to disulfoton from consumption of drinking water is likely to be negligible.

In the past, disulfoton was detected in some foods (see Section 5.5.4). However, the FDA has not detected disulfoton in the U.S. food supply in recent years (FDA 2017a, 2017b, 2018, 2019, 2022). The USGS estimated that <0.02 to 0.18 pounds of disulfoton per square mile were used on vegetable and fruit crops in two U.S. states in 2016 (USGS 2021). Use of disulfoton has substantially decreased since 2000, and no use of disulfoton on U.S. agriculture was reported after 2016, suggesting that use has ceased and it is not likely to be present in food grown in the United States.

Toxicokinetic data show that disulfoton is readily and extensively absorbed by the gastrointestinal tract. The urinary metabolites of disulfoton are DEP, DETP, DEDPT, and DEPTh. Although the occurrence of these phosphate esters in human urine may not result specifically from exposure to disulfoton, detection of these metabolites in human urine indicates the possibility of exposure to disulfoton or several other organophosphate insecticides. Using NHANES data from 1999 to 2008, Gillezeau et al. (2019) found that average urinary levels of organophosphate metabolites have decreased over time, but levels appear to have plateaued in recent years and some highly exposed individuals remain. NHANES data for organophosphate metabolites (from 2007–2008) showed specimens of urine collected contained detectable levels of DEP (detection limit of 0.37 µg/L) at a frequency of 31.40%, of DETP (detection limit of 0.56 µg/L) at a frequency of 41.12%, and of DEDTP (detection limit of 0.39 µg/L) at a frequency of 0.63% (Gillezeau et al. 2019). More recently, analysis of urine from NHANES for 2011–2012 showed detection (detection limit 0.1 ng/mL) of DETP at a frequency of 71% and DEDPT at a frequency of 5.4% of those tested (CDC 2019). However, as previously stated, these metabolites are not specific for disulfoton and can be found after exposure to other organophosphates.

5.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Workers involved in the manufacture, formulation, handling, or application of disulfoton, or those involved in the disposal of disulfoton-contaminated wastes are likely to be exposed to higher concentrations by dermal contact and inhalation than the general population. NIOSH (2018) recommends that the exposure level to skin not exceed 0.1 mg/m^3 for a 10-hour time-weighted average workday. In a study conducted by Storm et al. (2000), toxicity and other relevant data for disulfoton and 29 other organophosphate pesticides were evaluated to determine inhalation occupational exposure limits (OELs) and to support development of a risk assessment strategy for organophosphates in general. Specifically, the study assessed the value of relative potency analysis and the predictability of inhalation OELs by acute toxicity measures and by repeated oral exposure at the NOAEL. The OELs were derived by use of the endpoint of prevention of red blood cell AChE inhibition and by use of a weight-of-evidence risk assessment approach. When red blood cell AChE activity decreased to 70% (30% inhibition) of an individuals' baseline, it was concluded that the potential for overexposure to organophosphates exists and adverse effects may occur. It was advised that in cases where organophosphate workers experiencing this degree of red blood cell AChE inhibition occurs, actions be taken to prevent exposure until red blood cell AChE activity returns to the individuals' baseline. Suggested OEL values for the entire group of organophosphates evaluated ranged from 0.002 to 2 mg/m^3. The suggested OEL for disulfoton specifically was 0.01 mg/m^3. The suggested OEL for disulfoton was less than the current threshold limit value (TLV) of 0.1 mg/m^3 (Storm et al. 2000).

Older occupational studies showed that when Di-Syston, containing 75% disulfoton on a pumice granule, was applied to a field by air, the estimated inhalation exposure to disulfoton was 0.02 mg/8-hour day for the pilot and 0.03 mg/8-hour day for the ground staff (Myram and Forrest 1969). The estimated inhalation exposure to disulfoton for workers using ground machines was 0.33 mg/8-hour day (Myram and Forrest 1969).

Children may receive higher disulfoton doses from ingestion or dermal exposures if they play in contaminated soils near hazardous waste sites or in soils where a disulfoton pesticide was applied; however, this is less likely as disulfoton pesticides were cancelled in the United States in 2009. Previously allowed pesticides containing disulfoton may still be in circulation.