

5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

Disulfoton enters the environment primarily during its use as an insecticide/acaricide in crops and vegetables, and in homes and gardens. Other important pathways for disulfoton's entry into the environment are the disposal of liquid disulfoton wastes into soil evaporation pits, ditches, ponds (Winterlin et al. 1989), and hazardous waste sites. Thus, soil is the environmental medium most likely to be 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; Sanbom 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 (Cape1 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 10 °C river water was 7-41 days (Wanner et al. 1989). The measured whole-body bioconcentration factor 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; GreenhaIgh 1978; Harris et al. 1988; Jury et al. 1987a; Menzie 1972).

Disulfoton was detected at a maximum of 4.7 ng/m³ 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 µg/L 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.

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The Food and Drug Administration (FDA) estimated the average dietary intake of disulfoton from 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 1993b; FAO/WHO 1991; Winter 1992; Yess 1991).

Workers in industries that manufacture and formulate disulfoton, farm workers who enter treated fields after the insecticide has been applied and, in particular, applicators of the insecticide are at a higher risk of exposure than the general population. Among the general population, people who use the insecticide in homes and garden applications are at a higher risk of exposure, as are people who live near hazardous waste sites containing disulfoton.

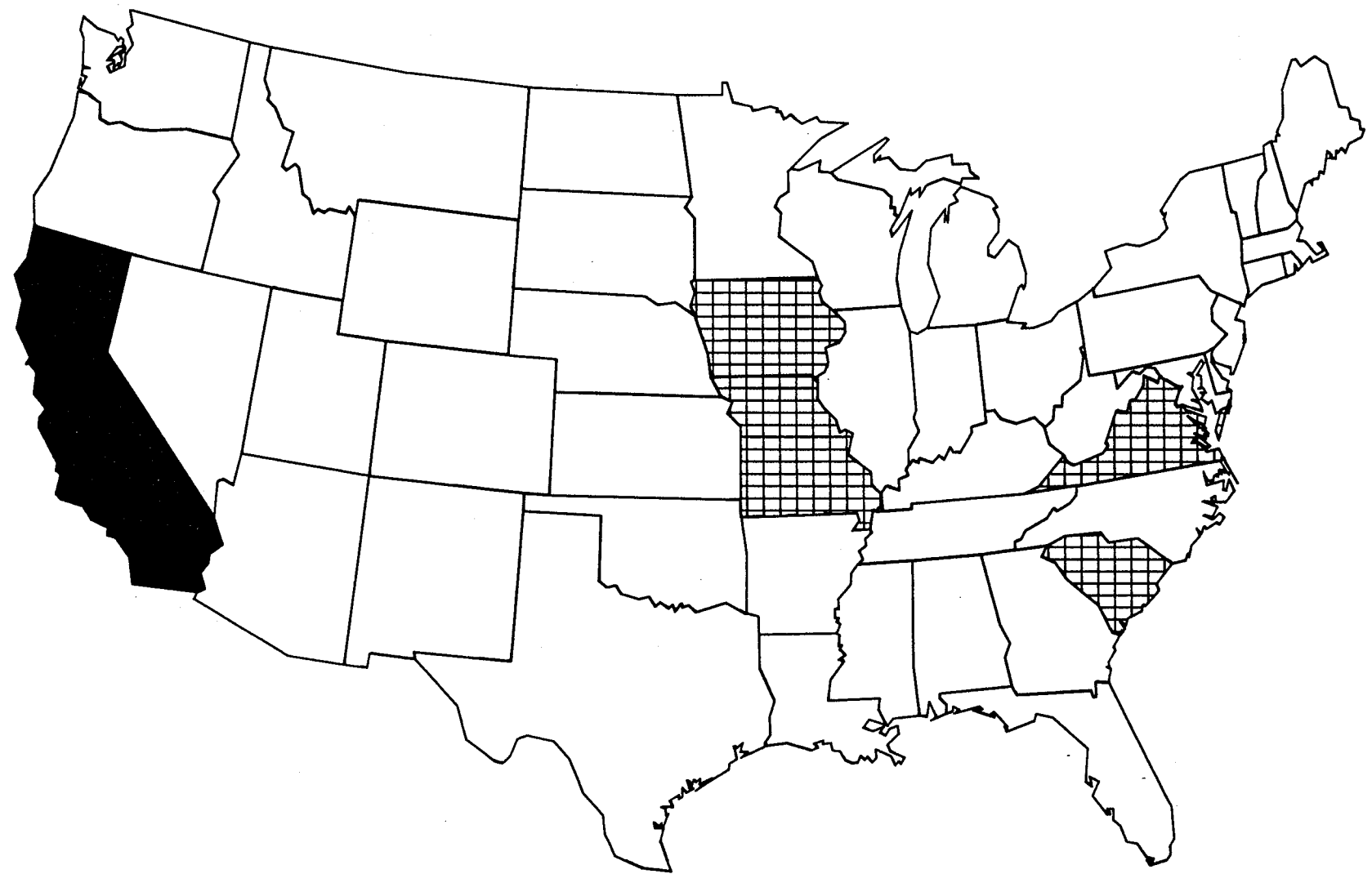
Disulfoton has been identified in 7 of the 1,408 hazardous waste sites on the NPL (HazDat 1994). The frequency of these sites within the United States can be seen in Figure 5-1.



5.2 RELEASES TO THE ENVIRONMENT

5.2.1 Air

Disulfoton may enter the atmosphere during its production and its use as an insecticide/acaricide. Air emissions from disulfoton production have been reported to be 0.5 kg per 1,000 kg (one metric ton) (Sittig 1980). Assuming a minimum production volume of 3.1 million pounds (1.4 million kg) per year, based on Gianessi and Puffer (1992) use estimates for 1989 through 1991, approximately 1,500 pounds (700 kg) are emitted to the atmosphere from production annually. Disulfoton is primarily released to the atmosphere during application of the granular insecticide by ground broadcast or aerial application (CPCR 1992). Disulfoton also enters the atmosphere when the emulsifiable insecticide is sprayed on crops and soil (CPCR 1992), and when the insecticide is aerially applied as a fog for controlling mosquitos (Racke 1992; Warnick and Eldredge 1972). Although not as significant a source, unintentional releases of disulfoton into the atmosphere include volatilization of applied insecticide from soil and plant surfaces, and accidental spills.

FIGURE 5-1. FREQUENCY OF NPL SITES WITH DISULFOTON CONTAMINATION *



FREQUENCY  1 SITE  3 SITES
*Derived from HazDat 1994

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5.2.2 Water

Other than aerial application over swamps for mosquito abatement, disulfoton is not known to be used over water. Potential sources of release into surface water include discharge of waste water from disulfoton manufacturing, formulation, and packaging facilities (HSDB 1994). Leaching and runoff from treated fields, pesticide disposal pits, or hazardous waste sites may contaminate both groundwater and surface water with disulfoton. Entry into water can also occur from accidental spills. Small amounts of volatilized disulfoton may be removed from the atmosphere as a result of wet deposition and may enter surface water (Racke 1992).

5.2.3 Soil

Disulfoton is 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 (HSDB 1994). Much of the foliar-applied disulfoton or its metabolites eventually reach the soil (Racke 1992). Soil in waste disposal sites may include manufacturing wastes containing disulfoton. A primary method for disposing of liquid pesticide wastes has been dumping liquid materials into soil evaporation pits, ditches, and ponds. Topsoil from such discharge areas can be expected to be contaminated with pesticides, as the soil from one such discharge pit contained disulfoton (Winterlin et al. 1989). Soil from tail water pits used for collecting irrigation runoff may also be a source of disulfoton if the soil has been treated with this insecticide (Kadoum and Mock 1978). Small amounts of disulfoton may enter soil by deposition of aerial disulfoton on land (Racke 1992). Entry may also occur from spills during storage, transport, or equipment loading and cleaning, although contemporary management practices significantly reduce the quantities of disulfoton accidentally released during its use (Racke 1992).

5.3 ENVIRONMENTAL FATE**5.3.1 Transport and Partitioning**

There is a paucity of experimental data regarding the transport and partitioning of disulfoton in air. Given the vapor pressure of 1.8×10^{-4} mm Hg at 20 °C (Melnikov 1971), disulfoton should exist almost entirely in the vapor phase in the atmosphere (Eisenreich et al. 1981). Because of low particle

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diameter, the removal rate by dry deposition is low for compounds that exist in the vapor phase in the atmosphere (Schroeder et al. 1987); therefore, depending on its reactivity characteristics, vapor-phase disulfoton may travel long distances in the air. The solubility of 25 mg/L (Sanborn et al. 1977) ensures that at least partial removal of atmospheric disulfoton will occur by wet deposition.

The transport of disulfoton from water to air can occur due to volatilization. Compounds with a Henry's law constant (H) of $<10\text{m}^{-5}\text{ atm}\cdot\text{m}^3/\text{mol}$ volatilize slowly from water (Thomas 1990). Therefore, disulfoton, with an H value of $2.17\times 10^{-6}\text{ atm}\cdot\text{m}^3/\text{mol}$ (Domine et al. 1992), will volatilize slowly from water. The rate of volatilization increases as the water temperature and ambient air flow rate 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 meters 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).

Adsorption to particulate matter will transport disulfoton from water to suspended solids and sediment in water. The estimated organic carbon-adjusted soil sorption coefficient (K_{oc}) for disulfoton varies between 600 and 1,603 (Jury et al. 1987a; Rao and Davidson 1982; Wauchope et al. 1992). This range of K_{oc} values suggests that disulfoton in water adsorbs moderately to suspended solids and sediments (Swann et al. 1983), and this process may transport considerable amounts of disulfoton from water to particulate matter.

Based on structure activity relationships, certain regression equations have been developed to estimate disulfoton's bioconcentration factor (BCF) from its water solubility and K_{oc} values. Based on these regression equations, the estimated BCF for disulfoton in aquatic organisms is 100-1 10 (Kenaga 1980). 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. 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

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≈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.

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 600-1,603 suggest that the adsorption of disulfoton to soil is moderately 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 (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 1989b). 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 can leach through certain soils into groundwater.

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 (Sanbom et al. 1977), is expected to be found mainly in runoff water. In a runoff event from agricultural soil in Nebraska, low levels of disulfoton

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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 (Sanbom et al. 1977). Recently, 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.

In the first year, the maximum concentrations of sulfoxide and sulfone in soil, seed potatoes, and foliage were approximately 2, 2, and 6 times, respectively, the concentrations of those metabolites measured in the second and third year treatments. These results demonstrated that enhanced microbial degradation of relatively minor insecticidal compounds in the soil can significantly affect insecticide levels in the plant (when these degradation products are the major insecticidal component accumulated). As the sulfoxide and the sulfone metabolites are the major toxicants in the foliage of potato plants grown in disulfoton-treated soil, this reduction in toxicant residues over time can be expected to reduce insecticide efficacy.

5.3.2 Transformation and Degradation

5.3.2.1 Air

One of the important reactions for most organic pollutants in the atmosphere is with hydroxyl radicals. Using an estimation method (Atkinson 1988), the estimated rate constant for the vapor-phase reaction of disulfoton with hydroxyl radicals is 13.2×10^{-11} cm³/molecule-sec. Based on this value and assuming an average atmospheric hydroxyl radical concentration of 5×10^5 radicals/cm³ (Atkinson 1988), the estimated half-life of disulfoton in the atmosphere due to this reaction is 3 hours. Therefore, disulfoton is short-lived in the atmosphere. 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. The importance of this reaction for the transformation of disulfoton cannot be assessed because a reaction rate is unavailable.

5.3.2.2 Water

The three processes responsible for the transformation and degradation of disulfoton in water are abiotic hydrolysis, photosensitized oxidation, and biodegradation. Disulfoton is most stable towards hydrolysis in the pH range 1-5 (Muhlmann and Schrader 1957). The experimentally estimated rate constants for the hydrolysis of disulfoton at 70 °C were 1.16×10^{-2} /hour, 1.56×10^{-2} /hour, 2.50×10^{-2} /hour, 3.22×10^{-2} /hour, and 9.61×10^{-2} /hour at pH of 5, 6, 7, 8, and 9, respectively (Muhlmann and Schrader 1957). The corresponding estimated half-lives are 60, 44, 27.6, 21.5, and 7.2 hours. The estimated hydrolysis half-life of disulfoton at 70 °C and pH 6.9 in ethanol-aqueous buffer solution (20:80) was 32 hours (Ruzicka et al. 1967). Other investigators reported the hydrolysis half-life of disulfoton under more realistic environmental conditions. For example, estimated hydrolysis half-lives were 103 days at 25 °C and pH 7 (Ellington et al. 1988) and 170 days at 11 °C and pH 7.9 (Wanner et al. 1989). Hydrolysis products of disulfoton are diethylthiophosphoric acid and 2-ethylmercaptiothio 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 (¹O₂) produced from the reaction of certain photochemically excited dissolved organic matters (e.g., humic and fulvic

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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 meters 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 μmol solution of disulfoton was exposed to October sunlight (Davis, California) in the presence of 100 μmol hydrogen peroxide, 49% of the insecticide disappeared in 10.2 days due to reaction with hydroxyl radicals (Draper and Crosby 1984). The rate of this reaction will be slower in natural surface water where the concentration of available hydroxyl radicals is lower than that available from the photolysis of 100 μmol hydrogen peroxide (levels ≤ 30 μmol hydrogen peroxide are generated in eutrophic 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 is expected to be important in water, 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 accident spill incident (Mossman et al. 1988); however, the removal of disulfoton by chemical processes was much slower than by biodegradation (Cape1 et al. 1988).

5.3.2.3 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 (see Section 5.3.2.2), 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

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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 (Bhaskaran et al. 1973). In flooded soil under anaerobic conditions, the reduction of disulfoton sulfoxide to disulfoton was due to biological conversion (Tomizawa 1975). The reduction of sulfone to either sulfoxide or sulfide (i.e., disulfoton) was not observed under the same conditions. 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 (Cape1 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 days 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 in soil varied between 28 and >64 days (Belanger and Hamilton 1979; Clapp et al. 1976; Jury et al. 1987b; Keamey 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.

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The presence of light and higher soil pH (pH 8 versus 5) also accelerated degradation of disulfoton in soil (Shaw 1975). The only metabolites isolated from disulfoton degradation in soil were the sulfoxide and sulfone (Chapman et al. 1994a, 1994b); only 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).

5.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT**5.4.1 Air**

Disulfoton was detected at maximum and mean concentrations of 4.7 and 0.1 ng/ m³ 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). No other air monitoring data were located.

5.4.2 Water

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). In an earlier survey of 48 municipal and 6 private groundwater supply systems in California, no disulfoton was detected at a minimum detection limit of 5 µg /L (Maddy et al. 1982). Disulfoton was not detected (detection limit of 1 µg /L) in over 1,508 samples from 358 groundwater wells in Wisconsin during 1980-1984 (Holden 1986; Krill and Sunzogni 1986). As of 1983, disulfoton was not detected (detection limit not given) in three samples of groundwater from Suffolk County, New York (Holden 1986). Disulfoton was detected in a 1985 survey of groundwater in the Nomini Creek Watershed in Virginia. The mean and maximum concentrations detected were 0.39 ppb and 2.87 ppb respectively. The frequency of detection of disulfoton in this watershed was 2.6% (Mostaghimi et al. 1993). Disulfoton was qualitatively detected (detection limit of 0.1 µg /L) in three groundwater samples from 1 of 479 hazardous waste disposal sites in the United States (Plumb 1991).

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Recently, levels of disulfoton in an agricultural watershed were monitored during a spring runoff event in Shell Creek, an eastern tributary of the Platte River in Nebraska. The concentrations of disulfoton in this agricultural watershed ranged from trace to 0.4 µg /L (Spalding and Snow 1989).

During 1981-1985, disulfoton and other pesticides were monitored at three watersheds adjacent to agricultural production areas: the mouths of the Grand River, the Saugeen River, and the Thames River, Ontario, Canada (Frank and Logan 1988). At a detection limit of <0.1 µg /L, no disulfoton was detected in the river water samples. Disulfoton was also not detected (detection limit of 0.003 µg /L) in samples of water collected from Lakes Superior and Huron, including Georgian Bay, in 1974 (Glooschenko et al. 1976).

5.4.3 Sediment and Soil

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). Disulfoton was detected in six of seven bottom soil samples from tail water pits used to collect irrigation runoff in Haskell County, Kansas, in 1974. The maximum and mean concentrations of disulfoton in these samples were 32.7 and 13.42 µg /kg respectively (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).

5.4.4 Other Environmental Media

Disulfoton was detected in 1 of 360 food composites collected from different locations in the United States in fiscal year 1969. The positive sample was a composite of leafy vegetables collected from Boston, Massachusetts, that contained 2 µg /kg of disulfoton (Comeliussen 1970). From this data, the estimated daily adult human intake of disulfoton was <1 µg /day in 1969 (Duggan and Comeliussen 1972). During fiscal year 1976, the FDA identified disulfoton in one sample each of cereal byproducts and leaf and stem vegetables, and in four samples of oilseed byproducts (Duggan et al. 1983). Disulfoton was also detected in one sample of whole grains for animal use and four miscellaneous animal feed samples (Duggan et al. 1983). The FDA's monitoring program for domestic and imported food commodities during fiscal years 1978-1982 detected disulfoton in

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unspecified foods at unspecified concentrations (Yess et al. 1991). During 1982-1986, the FDA Los Angeles District Laboratory analyzed 19,851 samples of domestic and imported food and feed commodities (Luke et al. 1988). Disulfoton sulfone was detected 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. In domestic foods, disulfoton sulfone was detected in broccoli, cabbage, Boston lettuce, butter lettuce, green leaf lettuce, iceberg lettuce, red leaf lettuce, romaine lettuce, potatoes, and spinach; it was also detected in imported tomatillos (Hundley et al. 1988). Disulfoton was not detected in 14,492 domestic and imported food samples analyzed as part of the FDA pesticide monitoring program for 1986-1987 (FDA 1988). Disulfoton was not detected in various domestic food commodities by state regulatory monitoring activities during fiscal year 1988-1989 (Minyard et al. 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) (Schattenburg and Hsu 1992). As part of the FDA's Pesticide Monitoring Program for domestic and imported foods, disulfoton residues have been detected in unspecified foods at unspecified concentrations during 1988-1989, 1989-1990, 199&1991, and 1991-1992 (FDA 1990, 1991, 1992, 1993).

Processing raw food reduced disulfoton levels in some potato products (Kleinschmidt 1971). Total residues were reduced by 35% with lye peeling. Lye peeling plus the first water blanching reduced the total disulfoton residue on a dry weight basis by 58, 74 and 61% for french fries, dehydrated potato cubes and dehydrated mashed potatoes, respectively.

In a study of pesticide residue contamination of processed milk-based and soy-based infant formula, disulfoton was not detected (detection limit <0.02 µg /g [ppm]) (Gelardi and Mountford 1993). The EPA Office of Water has recommended that disulfoton residues be monitored by states in their fish and shellfish contaminant monitoring programs in watersheds where this pesticide is currently used extensively in agriculture (EPA 1993a). While no fish or shellfish consumption advisories are currently in effect for disulfoton, this contaminant has not been widely monitored in national fish contaminant monitoring programs (e.g., the EPA National Study of Chemical Residues in Fish or the U.S. Fish and Wildlife Service National Contaminant Biomonitoring Program) (EPA 1993a).

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5.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

The general population is exposed to disulfoton by inhaling air and ingesting drinking water and food containing disulfoton. Toxicokinetic data show that disulfoton is readily and extensively absorbed by the gastrointestinal tract (see Section 2.4). The urinary metabolites of disulfoton are DEP, DETP, DEDPT, and DEPTH (see Section 2.3). 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. Specimens of urine collected at 64 locations across the United States, comprising the sample areas in NHANES II, reported detection (detection limit 20 µg /L or ppb) of DEP and DETP at a frequency of 6-7% and DEDPT at a frequency of <1% of those tested (Murphy et al. 1983). Other than during home and garden insecticide application, exposure of the general public to disulfoton from skin contact with consumer products is not expected. Those among the general population who use the insecticide for homes and gardens are potentially at higher risk of exposure to disulfoton. It has been estimated that application to rose plants and other ornamentals constitute by far the largest home and garden use of disulfoton (Whitmore et al. 1992).

Disulfoton has been very infrequently detected in ambient air (0.8% of air samples from 10 cities) and at very low concentrations (mean of 0.1 ng/ m³ in detectable samples) (see Section 5.4.1). 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 5.4.2), possibly because of it occurs at very low concentrations and has only infrequently been detected in groundwater (Cohen 1986; Hallberg 1989; Holden 1986; Mostaghimi et al. 1993; Plumb 1991). Therefore, general population exposure to disulfoton from consumption of drinking water is negligible.

Disulfoton has been detected in some foods (see Section 5.4.4); thus, consumers can be exposed to disulfoton by ingesting contaminated food. The FDA has estimated daily food intakes of disulfoton for different age/sex groups in the United States for fiscal years 1982-1984 as follows: 6-11 months = 0.6; 2 years = 1.1; 14-16-year-old females = 0.4; 14-16-year-old males = 0.3; 25-30-year-old females = 0.3; 25-30-year-old males = 0.3; 60-65-year-old females = 0.5; and 60-65-year-old males = 0.6 (ng/kg body weight/day) (Gunderson 1988). The reported corresponding values for combined disulfoton and its metabolites (e.g., sulfone, sulfoxide, etc.) are 2.2, 4.8, 2.3, 1.8, 2.4, 2.2, 2.9, and 3.0 (ng/kg body weight/day) (Gunderson 1988). The FDA reported the following

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dietary intakes of disulfoton for different age/sex groups in fiscal year 1990: 6-11 months, <0.1; 14-16-year-old males, 0.2; and 60-65-year-old females, 0.3 (ng/kg body weight/day) (FDA 1991, Winter 1992; Yess 1991). Most recently, the FDA reported the following dietary intakes of disulfoton (total) for different age/sex groups averaged over 1986-1991: 6-11 months, 0.1; 2 years, 0.3; 14-16-year-old males, 0.2; 14-16-year-old females, 0.2; 25-30-year-old females, 0.4; 25-30-year-old males, 0.3; 60-65-year-old females, 0.4; and 60-65-year-old males, 0.3 (ng/kg body weight/day) (FDA 1993). These 1982-1984, 1990, and 1986-1991 intakes estimated from the FDA Total Diet Study Program show a declining trend in the estimated daily food intakes of disulfoton over the past 10 years. All of the FDA estimated daily food intakes for the various sex and age groups are more than 1,000 times lower than the FAO/WHO Acceptable Daily Intake (ADI) value of 2,000 ng/kg body weight/day and an EPA revised reference dose of 40 ng/kg body weight/day ((FAO/WHO 1991; FDA 1991, 1993; Winter 1992; Yess 1991).

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. When granular disulfoton was applied at 0.75 active ingredient per acre 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).

No information was found in the National Occupational Exposure Survey (NOES) conducted by NIOSH from 1981 to 1983 on the number of workers and the number of facilities where workers could be potentially exposed to disulfoton in the United States (NOES 1990). NIOSH (1992) recommends that the exposure level to skin not exceed 0.1 mg/ m³ for a 10-hour time weighted average workday.

5.6 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Other than individuals who are occupationally exposed to disulfoton during its production, formulation, packaging, distribution, use, or disposal, populations exposed to higher than background concentrations of disulfoton in ambient air include those living near chemical manufacturing or processing sites, individuals living on farms or in the vicinity of agricultural areas where disulfoton is extensively used,

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and individuals living near hazardous waste sites. Individuals living near these sites may also be exposed to potentially higher concentrations of disulfoton or its metabolites in their drinking water if they obtain tap water from wells near these sources. Children may receive higher disulfoton doses from dermal exposures if they play on freshly treated soils. In addition, children may receive potentially higher oral doses from ingestion of disulfoton contaminated soils from their hands while playing in contaminated areas.

5.7 ADEQUACY OF THE DATABASE

Section 104(i)(5) of CERCLA, as amended, directs the Administrator of ATSDR (in consultation with the Administrator of EPA and agencies and programs of the Public Health Service) to assess whether adequate information on the health effects of disulfoton is available. Where adequate information is not available, ATSDR, in conjunction with the NTP, is required to assure the initiation of a program of research designed to determine the health effects (and techniques for developing methods to determine such health effects) of disulfoton.

The following categories of possible data needs have been identified by a joint team of scientists from ATSDR, NTP, and EPA. They are defined as substance-specific informational needs that if met would reduce the uncertainties of human health assessment. This definition should not be interpreted to mean that all data needs discussed in this section must be filled. In the future, the identified data needs will be evaluated and prioritized, and a substance-specific research agenda will be proposed.

5.7.1 Identification of Data Needs

Physical and Chemical Properties. As seen in Table 3-2, the relevant physical and chemical properties of disulfoton are known (Bowman and Sans 1983; Domine et al. 1992; HSDB 1994; Kenaga and Goring 1980; Melnikov 1971; Merck 1989; Sanbom et al. 1977; Worthing 1987) and predicting the environmental fate and transport of disulfoton based on K_{ow} , K_{oc} , and H is possible.

Production, Import/Export, Use, Release, and Disposal. Knowledge of production and use data for a chemical is important in predicting its potential for environmental contamination and human exposure. Since disulfoton is produced by only one manufacturer (SRI 1994), to maintain confidentiality, its recent production volume is not known. Similarly, data concerning the import and

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export volumes for disulfoton in recent years are not available. Estimates of its yearly use in the United States are available (Gianessi 1986; Gianessi and Puffer 1992; Goodrich et al. 1991; VonRumker et al. 1974). No information in the available literature was located that indicates disulfoton's use in any consumer products other than edible crops and vegetables during and after their planting (EPA 1984b; VonRumker et al. 1974). Disulfoton enters the environment primarily during its use as an agricultural insecticide/acaricide on a variety of field and vegetable crops and in its use in home and garden applications. Disulfoton wastes have been disposed of in soil evaporation pits, ditches, and ponds (Winterlin et al. 1989) and in hazardous waste sites. Thus, soils are most likely to be contaminated with this pesticide. Although some information regarding the disposal of wastes containing disulfoton is available (Berlow and Cunningham 1989; HSDB 1994; IRPTC 1985; Sittig 1980), more detailed information on the method used for the disposal of aqueous wastes would be helpful. The standards promulgated by the EPA for the disposal of wastes containing disulfoton are available (Berlow and Cunningham 1989).

According to the Emergency Planning and Community Right-to-Know Act of 1986, 42 U.S.C. Section 11023, industries are required to submit chemical release and off-site transfer information to the EPA. The Toxics Release Inventory (TRI), which contains this information for 1988-1992, became available in May of 1994. This database will be updated yearly and should provide a list of industrial facilities and emissions. However, no TRI data were located for disulfoton because this chemical is not required to be reported by chemical producers (EPA 199313).

Environmental Fate. Information regarding the fate of disulfoton in the air was not located in the literature. Although the available data indicate that the concentration of disulfoton in air will be low (Carey and Kutz 1985), more information would help predict the distance of its aerial transport. The fate of disulfoton in water is better studied (Wanner et al. 1989). Although it has been estimated that sorption onto particulates and settling into the sediment may not be important for disulfoton in Rhine River water, more information regarding the relative importance of sorption for disulfoton's removal from water to sediment would be helpful (Wanner et al. 1989). There is conflicting evidence in the literature (Harris 1969; Helling et al. 1974; King and McCarty 1968; McCarty and King 1966) regarding disulfoton's mobility in soil. Disulfoton is degraded in soil by hydrolysis, photoinduced oxidation, and biodegradation (Bhaskaran et al. 1973; Cape1 et al. 1988; Chapman et al. 1993, 1994b; Gohre and Miller 1986; Hebert and Miller 1990; Miller et al. 1989; Tomizawa 1975; Wanner et al.

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1989). Additional information on degradation of disulfoton in water and air and the fate of the degradation products in soil would be helpful.

Bioavailability from Environmental Media. Available information regarding the rate of disulfoton absorption following inhalation, oral, or dermal contact has been discussed in the Toxicokinetics section (see Section 2.3). Although no data on disulfoton's bioavailability from contaminated air are available, the bioavailability from inhalation exposure is expected to be high because disulfoton is likely to be present in the vapor phase (Eisenreich et al. 1981) and not in the particulate phase in the adsorbed state. Similarly, no data on the bioavailability of disulfoton from water and soil or plant material are available; however, disulfoton adsorbs rather strongly to soil (Harris 1969; Helling et al. 1974; Wauchope et al. 1992). Since the part that remains adsorbed to soil or sediments may, at most, be partially bioavailable, disulfoton is expected to have reduced bioavailability from soil and water. Data on the bioavailability of disulfoton from actual environmental media need further development.

Food Chain Bioaccumulation. Disulfoton bioconcentrates to a moderate extent in fish (Takase and Oyama 1985; Tomizawa 1980); however, measured bioconcentration factor (BCF) values are not available for a large number of edible invertebrate and fish species. EPA (1993a) has recently recommended that states monitor disulfoton in fish and shellfish in watersheds where it is extensively used, and additional information on disulfoton residues may be available in the future. There is also a lack of data in the literature regarding the biomagnification potential of disulfoton through aquatic food chains. Available data on terrestrial food chains indicate that disulfoton is translocated from the root to aerial parts of the plants, where it is quickly metabolized to sulfone and sulfoxide (Nash 1974; Szeto et al. 1983a, 1983b). Data regarding the biomagnification potential of disulfoton in terrestrial and aquatic food chains would be desirable. These data would be helpful in assessing the potential for human exposure as a result of ingestion of contaminated food.

Exposure Levels in Environmental Media. Although some data on the levels of disulfoton in ambient air are available (Carey and Kutz 1985), these data are neither current nor general enough to estimate inhalation exposure to disulfoton for the general population in the United States. No data on the level of disulfoton in drinking water were located in the literature, although disulfoton has been detected in groundwater (Cohen 1986; Hallberg 1989; Holden 1986; Mostaghimi et al. 1993). More recent data regarding the levels of disulfoton in ambient air, in drinking water, and in soil are needed.

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Data on disulfoton levels in food and recent estimates of human intake of disulfoton from foods are available (Duggan et al. 1983; FDA 1990, 1991, 1992, 1993; Gelardi and Mountford 1993; Gunderson 1988; Luke et al. 1988; Schattenburg and Hsu 1992; Yess 1991).

Reliable monitoring data for the levels of disulfoton in contaminated media at hazardous waste sites are needed so that the information obtained on levels of disulfoton in the environment can be used in combination with the known body burden of disulfoton to assess the potential risk of adverse health effects in populations living in the vicinity of hazardous waste sites.

Exposure Levels in Humans. No data on disulfoton levels in various human tissues and body fluids of a control population, populations near hazardous waste sites, or occupationally exposed groups in the United States are available. The levels of disulfoton metabolites (DEP [0.05 ppm], DETP [0.04 ppm], DEPTH [0.005 ppm], dimethyl phosphate [0.04 ppm], dimethyl thiophosphate [0.180 ppm], and dimethyl phosphorothiolate [0.004 ppm]) in the urine of disulfoton formulators have been measured (Brokopp et al. 1981). Data on the levels of disulfoton and its metabolites in body tissues and fluids are needed to estimate the extent of exposure to disulfoton.

This information is necessary for assessing the need to conduct health studies on these populations.

Exposure Registries. No exposure registries for disulfoton were located. This substance is not currently one of the compounds for which a subregistry has been established in the National Exposure Registry. The substance will be considered in the future when chemical selection is made for subregistries to be established. The information that is amassed in the National Exposure Registry facilitates the epidemiological research needed to assess adverse health outcomes that may be related to exposure to this substance.

5.7.2 Ongoing Studies

A search of federal research programs in progress indicates a few projects that would fill some of the existing data gaps. Drs. Singmaster and Acin-Diaz of the University of Puerto Rico are developing methods for determining disulfoton residue levels in food commodities and for disulfoton disposal. Dr. Brady of the University of Georgia is conducting a study to determine disulfoton's dissipation on peach fruit, on foliage, in soil, in tank mixes, and in building interiors.