

## 6. POTENTIAL FOR HUMAN EXPOSURE

### 6.1 OVERVIEW

Diazinon has been identified in at least 25 of the 1,699 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HazDat 2008). However, the number of sites evaluated for diazinon is not known. The frequency of these sites can be seen in Figure 6-1. Of these sites, all are located within the United States.

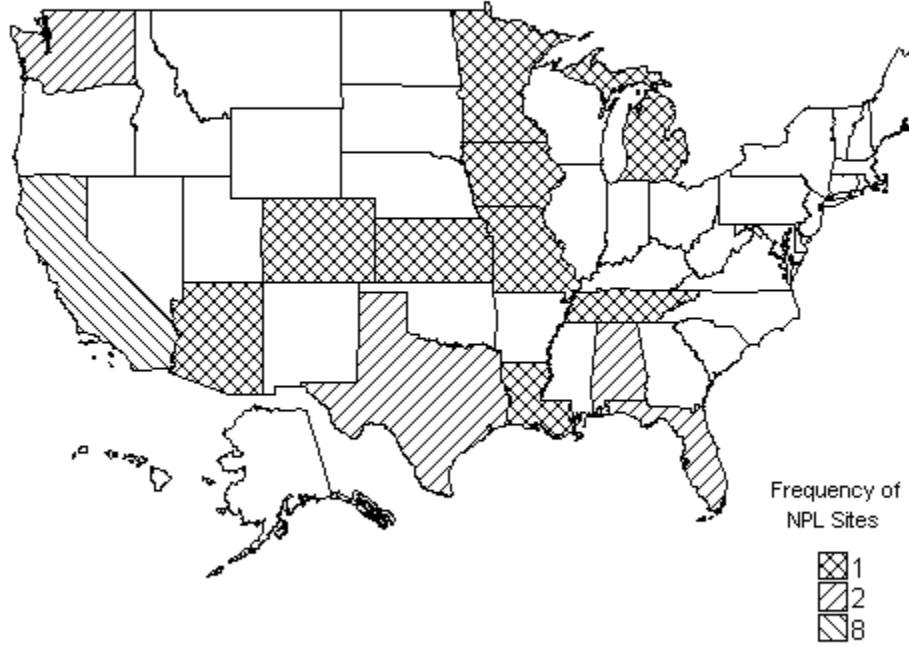
Diazinon is released to the environment solely by human activities. Major atmospheric emissions result from volatilization of the chemical from soil resulting from its extensive use as an insecticide or from drift during pesticide application. Diazinon is released to surface waters directly by point source discharges, from drift during pesticide applications, and by runoff from agricultural and urban areas (EPA 1995a, 1995b).

Diazinon is found in all environmental compartments, but shows no pronounced tendency to partition to a particular environmental medium. Given adequate time, diazinon will be degraded by abiotic and biotic processes so that the parent compound is not persistent. Degradation products of diazinon include diazoxon, a toxic degradate, and 2-isopropyl-6-methyl-4-hydroxypyrimidine (IMHP or oxypyrimidine), a persistent, less toxic degradate (EPA 2004b). Oxypyrimidine is the main soil and water degradate of diazinon (EPA 2004b). Diazinon has been detected in the atmosphere and trace amounts of its oxidation product (diazoxon) have also been detected. The diazoxon to diazinon ratio ranged from 0.056 to 7.1, but was generally <0.4 (Glotfelty et al. 1990a). In a study of diazinon use in the Central Valley of California, Seiber et al. (1993) reported that during daylight hours, the oxon to thion ratio in the atmosphere averaged 0.52, while at night, the ratio was 0.10. Diazinon can be converted to diazoxon in the atmosphere via ultraviolet (UV) radiation (Seiber et al. 1993). The estimated half-life for the vapor phase reaction of diazinon with hydroxyl radicals is approximately 4 hours (Meylan and Howard 1993). Diazinon can be transported moderate distances in the air from its original point of use (Zabik and Seiber 1993).

Diazinon released to surface waters or soil is subject to volatilization, photolysis, hydrolysis, and biodegradation. Biodegradation, primarily under aerobic conditions, is a major fate process for diazinon associated with water and soil. Diazinon can be biodegraded under anaerobic conditions as well. Hydrolysis is an important mechanism for degradation, particularly at low pH in water and soil. Diazinon has a relatively short half-life in water, ranging from 70 hours to 12 weeks depending on pH, temperature,

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**Figure 6-1. Frequency of NPL Sites with Diazinon Contamination**



Derived from HazDat 2008

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and sunlight as well as the presence of microorganisms (Chapman and Cole 1982; EPA 1976; Ferrando et al. 1992; Frank et al. 1991b; Scheunert et al. 1993; Schoen and Winterlin 1987; Sharom et al. 1980b). The half-life of diazinon in soil is influenced by the pH conditions in the soil and the soil type. The half-life values at pH 4, 7, and 10 were 66, 209, and 153 days, respectively, in sandy loam; 49, 124, and 90 days, respectively, in clay loam; and 14, 45, and 64 days, respectively, in sandy loam amended with peat (Schoen and Winterlin 1987). Diazinon is moderately mobile in some soils, particularly those with an organic matter content <3%, and can leach from soil into groundwater. If released to water, this pesticide does not bioaccumulate (bioconcentration factors [BCFs] values generally <100) in aquatic organisms.

In the United States, monitoring efforts under many national programs have not analyzed for this chemical. Diazinon has been identified in air samples from both rural and urban areas and in indoor air in both domestic and commercial buildings. It has also been detected in surface water, effluents from publicly owned treatment works (POTWs), and groundwater. It has been detected in soil and sediment in areas where it is extensively used in agriculture. Current information is lacking on the total amount of diazinon released to the environment and on the amount of diazinon that partitions into each environmental compartment.

The best-documented concern over diazinon relates to acute exposures of humans during or immediately following pesticide applications. This concern is warranted, since diazinon has been widely used, with many applications in urban areas (homes and gardens) that may have increased the possibilities of human exposure. In order to mitigate the exposure and risk to the general population, especially children, the EPA has phased out all residential uses of diazinon as of December 2004 (EPA 2004b). Diazinon and its major metabolite, diazoxon, have significant acute toxicity to humans. General population exposure to diazinon may occur through ingestion of contaminated food or drinking water and inhalation. Ingestion of foods contaminated with small residues of diazinon is the most likely route of exposure for the general population not living in areas where diazinon is extensively used. The general population may also be exposed to diazinon through inhalation of contaminated ambient (outdoor) air.

Populations living within or very near areas of heavy agricultural diazinon use would have an increased risk of exposure to relatively larger amounts of diazinon through dermal contact with contaminated plants, soils, surface waters, or artificial surfaces such as playground equipment and pavements; by inhalation of the mist formed from the applied insecticide; or by ingestion of water or food-borne residues. Those likely to receive the highest levels of exposure are those who are involved in the

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production, formulation, handling, and application of diazinon, farm workers who enter treated fields prior to the passage of the appropriate restricted entry intervals, and workers involved in the disposal of diazinon or diazinon-containing wastes. Dermal contact appears to be the major route of exposure for workers. Inhalation of diazinon in occupational settings depends on its volatility, the type of formulation used, and the application technique employed.

**6.2 RELEASES TO THE ENVIRONMENT**

Diazinon has been released to the environment mainly as a result of its extensive use as an insecticide for household lawn and garden pest control, indoor residential crack and crevice treatments and pest collars, and agricultural pest control. In order to reduce exposure to children and others, a December 2000 agreement began a phase out of residential uses of diazinon, which was completed in December 2004. Future releases of diazinon will mainly be a result of agricultural use by aerial and ground spraying and spreading. For 1987 through 1997, total annual domestic usage of diazinon was over 13 million pounds. Approximately 4 million pounds of active ingredient diazinon are used annually on agricultural sites (EPA 2004b). There are no known natural sources of the compound. Diazinon has been identified in at least 25 of the 1,699 hazardous waste sites on the NPL (HazDat 2008).

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 or more 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  $\geq 25,000$  pounds of any TRI chemical or otherwise uses  $>10,000$  pounds of a TRI chemical in a calendar year (EPA 2005).

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**6.2.1 Air**

Estimated releases of 358 pounds (~0.16 metric tons) of diazinon to the atmosphere from 21 domestic manufacturing and processing facilities in 2005, accounted for about 1.5% of the estimated total environmental releases from facilities required to report to the TRI (TRI05 2007). These releases are summarized in Table 6-1.

Diazinon is released into the atmosphere solely by human activities associated with its production and use as an insecticide. These releases include releases to ambient air from production and from agricultural applications. Releases have also resulted from domestic lawn and garden applications, and releases to indoor air from pest-control treatment of domestic and commercial buildings. It appears that diazinon that has been applied to a field can undergo volatilization to the atmosphere (Glotfelty et al. 1990a; Schomburg et al. 1991; Seiber et al. 1993; Zabik and Seiber 1993). Glotfelty et al. (1990b) estimated that up to 24% of the diazinon applied to dormant peach orchards may be released through long-term volatilization losses even though volatilization quickly declines to low levels. Before residential use was cancelled in 2004 (EPA 2004b), home and garden application once accounted for over 40% of total diazinon usage; it is not possible to estimate volatilization from these applications.

Diazinon was detected in air at 1 of the 1,699 current or former NPL sites where diazinon has been identified in some environmental medium (HazDat 2008).

**6.2.2 Water**

Estimated releases of 10,287 pounds (~4.67 metric tons) of diazinon to surface water and to publicly owned treatment works (POTWs) from 21 domestic manufacturing and processing facilities in 2005, accounted for about 43% of the estimated total environmental releases from facilities required to report to the TRI (TRI05 2007). These releases are summarized in Table 6-1.

Diazinon is released into water directly from point source discharges, from drift during pesticide applications, and from nonpoint-source runoff from agricultural and urban areas. The use of permit compliance bioassay testing has helped identify point source discharges with acutely toxic effluents, and follow-up chemical analyses have pinpointed the identity of specific toxicants (Amato et al. 1992). Such work has led to the identification of diazinon as a cause of toxicity in POTW discharges (Amato et al. 1992; Burkhard and Jenson 1993). This is not surprising given the former widespread use of diazinon in

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**Table 6-1. Releases to the Environment from Facilities that Produce, Process, or Use Diazinon<sup>a</sup>**

State <sup>c</sup>	RF <sup>d</sup>	Reported amounts released in pounds per year <sup>b</sup>					Total release		
		Air <sup>e</sup>	Water <sup>f</sup>	UI <sup>g</sup>	Land <sup>h</sup>	Other <sup>i</sup>	On-site <sup>j</sup>	Off-site <sup>k</sup>	On- and off-site
AR	2	0	No data	0	0	0	0	0	0
AZ	1	0	No data	0	0	0	0	0	0
CA	1	0	No data	0	0	0	0	0	0
GA	2	9	5	0	0	0	14	0	14
KS	1	10	No data	0	0	0	10	0	10
LA	1	0	0	0	0	0	0	0	0
MO	1	0	No data	0	0	0	0	0	0
NE	1	0	No data	0	0	0	0	0	0
OH	2	250	0	0	0	0	250	0	250
TX	3	11	5,841	0	0	17	5,852	17	5,869
WY	1	250	No data	0	250	3	500	3	503
Total	16	530	5,846	0	250	20	6,626	20	6,646

<sup>a</sup>The TRI data should be used with caution since only certain types of facilities are required to report. No data. This is not an exhaustive list. Data are rounded to nearest whole number.

<sup>b</sup>Data in TRI are maximum amounts released by each facility.

<sup>c</sup>Post office state abbreviations are used.

<sup>d</sup>Number of reporting facilities.

<sup>e</sup>The sum of fugitive and point source releases are included in releases to air by a given facility.

<sup>f</sup>Surface water discharges, waste water treatment-(metals only), and publicly owned treatment works (POTWs) (metal and metal compounds).

<sup>g</sup>Class I wells, Class II-V wells, and underground injection.

<sup>h</sup>Resource Conservation and Recovery Act (RCRA) subtitle C landfills; other on-site landfills, land treatment, surface impoundments, other land disposal, other landfills.

<sup>i</sup>Storage only, solidification/stabilization (metals only), other off-site management, transfers to waste broker for disposal, unknown.

<sup>j</sup>The sum of all releases of the chemical to air, land, water, and underground injection wells.

<sup>k</sup>Total amount of chemical transferred off-site, including to POTWs.

RF = reporting facilities; UI = underground injection

Source: TRI05 2007 (Data are from 2005)

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urban areas to control indoor pests and lawn and garden pests. It is easy for diazinon and its residues to reach the sewer collection systems for many POTWs.

In addition to loadings passing through sewage treatment systems, diazinon can reach surface waters directly from point source discharges (Braun and Frank 1980), from nonpoint-source inputs introduced from agricultural (Braun and Frank 1980; Kendall et al. 1993; Maguire and Tkacz 1993; Szeto et al. 1990; USGS 1993; Wan et al. 1994), or from suburban runoff (Frank et al. 1991b). It is impossible to obtain estimates of these loadings to surface waters. Water concentrations and transport of diazinon through the Sacramento-San Joaquin Delta and the adjacent portions of San Francisco Bay were studied in 1993 by the U.S. Geological Survey (USGS 1993). Diazinon was applied as a dormant spray in the Central Valley of California during 2 weeks of dry weather in January 1993. Pulses of elevated diazinon concentrations were detected in the Sacramento and San Joaquin Rivers after a series of rainstorms in early February 1993. All concentrations of diazinon measured in river and bay water samples exceeded 9 ng/L. Contaminated water samples collected from the San Joaquin River produced 100% mortality in bioassay tests conducted with *Ceriodaphnia dubia* for 12 consecutive days from February 8 to 19. The mortality of this sensitive indicator species was attributed to agricultural runoff of diazinon associated with the February rain events (USGS 1993). Banks et al. (2005) collected 1,243 surface water samples at 70 monitoring stations from rural and urban streams in Denton, Texas during the years of 2001–2004 and monitored for diazinon before and after the EPA ban on its residential uses. The total number of samples having diazinon concentrations above the lower limits of detection significantly decreased between 2001 and 2004, with the average diazinon concentration falling from 2.58 to 0.85 µg/L. These results indicate that the phasing out of residential uses of diazinon has led to a significant decrease in surface water occurrences (Banks et al. 2005).

Since diazinon is moderately mobile in soils under certain conditions, it has the potential to migrate through the soil and into groundwater. Detections have been made in some groundwater wells in the United States (Cohen 1986; EPA 1989). In areas with heavy applications of diazinon combined with irrigation or water-level adjustment techniques, diazinon detections in groundwater also have been documented (Cohen 1986; Frank et al. 1987, 1990b). It has not been possible to obtain quantifiable estimates of these diazinon loadings to groundwater.

Diazinon has been detected in surface water at 5 of the 1,699 current or former NPL sites and in groundwater at 8 of the 1,699 current or former NPL sites where diazinon has been identified in some environmental medium (HazDat 2008).

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**6.2.3 Soil**

Estimated releases of 13,123 pounds (~5.95 metric tons) of diazinon to soils from 21 domestic manufacturing and processing facilities in 2005, accounted for about 55% of the estimated total environmental releases from facilities required to report to the TRI (TRI05 2007). No additional environmental releases via underground injection were reported (TRI05 2007). These releases are summarized in Table 6-1.

Diazinon is released into soils primarily from its registered use on various agricultural crops and its former use in home garden and lawn applications. Soils are the target for the vast majority of diazinon applications both as a nematocide and as an insecticide agent. In agricultural areas, diazinon may also be transferred to aquatic sediments (Domagalski and Kuivila 1993; Szeto et al. 1990; Wan et al. 1994). Since diazinon undergoes various activation and degradation reactions in the course of time ranging from hours to months, these loadings to soils and sediments are a temporary phenomena.

Diazinon has been detected in soil at 9 of the 1,699 current or former NPL sites and in sediment at 4 of the 1,699 current or former NPL sites where diazinon has been identified in some environmental medium (HazDat 2008).

**6.3 ENVIRONMENTAL FATE**

Diazinon can move into various environmental compartments, but there does not appear to be a major reservoir or sink for this chemical in any specific environmental compartment primarily because of its relatively rapid degradation in each environmental medium.

**6.3.1 Transport and Partitioning**

Based on its vapor pressure (see Table 4-2), if diazinon is released to the atmosphere, it will be expected to exist both in the vapor phase and particulate phase (Eisenreich et al. 1981). Glotfelty et al. (1990a) reported that during stagnant inversion fog events in the Central Valley of California, 56 and 19% of the diazinon in the air-phase was associated with vapor and aerosol particles, respectively, and only 24% of the diazinon was dissolved in the water phase. Schomburg et al. (1991) reported slightly different distributions for fog events resulting from advected oceanic fog. In this study, 26 and 10% of the diazinon in the air-phase was associated with vapor and aerosol particles, respectively; 62% of the



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diazinon was dissolved in the water phase. Zabik and Seiber (1993) studied the atmospheric transport of diazinon from California's Central Valley to the Sierra Nevada Mountains. These samples collected during January through February 1991 represented the simultaneous collection of both vapor and particulate phases. Concentrations of diazinon and diazoxon were 13–10,000 and 4–3,000 pg/m<sup>3</sup>, respectively, for samples collected at the 114 m elevation and 1.4–12 and 1.8–13 pg/m<sup>3</sup>, respectively, at the 533 m elevation. The pesticide concentrations in air samples decreased with distance and elevation moving east from the Central Valley into the higher elevations of the Sierra Nevada Mountains. At times, air concentrations at the 114 m elevation were 1,000 times greater than concentrations detected at 533 m elevation. Concentrations at the 1,920 m elevation were typically below the limit of quantification. Wet deposition samples collected at the 114 m elevation contained up to 6,100 pg/mL diazinon and 2,300 pg/mL diazoxon.

Limited data based on atmospheric sampling and laboratory studies (Glotfelty et al. 1990a, 1990b) suggest a much greater potential for diazinon transport into the atmosphere after application to soils and vegetation. While the activation process (diazinon to diazoxon conversion) in the air would tend to transform diazinon fairly rapidly, the possibility of atmospheric transport means that this pesticide can move some distance from agricultural to nonagricultural areas (Glotfelty et al. 1990a, 1990b; Schomburg et al. 1991; Seiber et al. 1993; Zabik and Seiber 1993).

Diazinon released to water from both point and nonpoint sources may be emitted to the atmosphere by volatilization, sorbed to soils and sediments, or accumulated in aquatic organisms. While volatilization of diazinon may not be expected to be significant based upon the Henry's law constant (see Table 4-2), it can be an important transport process. Sanders and Seiber (1983) reported that 17% of the diazinon added to a model pond volatilized in 24 hours. Diazinon released to water also may be adsorbed moderately by soils and sediments based on its organic carbon partition coefficient ( $K_{oc}$ ) values measured in soil (Sharom et al. 1980a). Because this pesticide is only moderately adsorbed by some soils, leaching into groundwater can occur.

Diazinon does not significantly bioaccumulate in aquatic organisms. A comparison of BCF values obtained for various freshwater and saltwater fish and invertebrate species is presented in Table 6-2. The BCF values generally range from 4 to 337, but there are only a few cases where the measured BCF value for diazinon exceeds 100. In those experiments where testing was continued for several days after exposure to the diazinon had ended, tissue residues generally decreased rapidly within 1–5 days (El Arab et al. 1990; Sancho et al. 1993; Tsuda et al. 1989, 1990, 1995). Despite the fairly low BCF values, some

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**Table 6-2. Bioconcentration Data for Diazinon**

Species common name/ Scientific name	Exposure type	Duration (days)	BCF <sup>a</sup>	Reference
Freshwater				
Shrimp/ <i>Paratya compressa</i> <i>compressa</i>	F	3	4	Seguchi and Asaka 1981
Oriental weatherfish/ <i>Misgurnus</i> <i>anguilli candatus</i>	F	14	28	Seguchi and Asaka 1981
Common carp/ <i>Cyprinus carpio</i>	F	3	130	Seguchi and Asaka 1981
Perch/ <i>Sarotherodon galilaeus</i>	S	3	39	El Arab et al. 1990
Rainbow trout/ <i>Oncorhynchus</i> <i>mykiss</i>	F	3	92	Seguchi and Asaka 1981
Brook trout/ <i>Salvelinus fontinalis</i>	F	210	25	EPA 1977
Guppy/ <i>Poecilia reticulata</i>	R	2	39	Keizer et al. 1991
Zebra fish/ <i>Brachydanio rerio</i>	R	2	300	Keizer et al. 1991
Willow Shiner/ <i>Gnathopogon</i> <i>caerulescens</i>	F	7	248	Tsuda et al. 1989
Killifish/ <i>Oryzias latipes</i>	F	3	20	Tsuda et al. 1995
Killifish/ <i>O. latipes</i>	F	7	94	Tsuda et al. 1997
Fathead minnow/ <i>Pimephales</i> <i>promelas</i>	F	2B304	337 <sup>b</sup>	Veith and Kosian 1983
Goldfish/ <i>Carassius auratus</i>	F	7	49	Tsuda et al. 1997
White cloud mountain fish/ <i>Tanichthys albonubes</i>	F	7	36	Tsuda et al. 1997
Saltwater				
Sheepshead minnow/ <i>Cyprinodon</i> <i>variegatus</i>	F	108	147	Goodman et al. 1979

<sup>a</sup>BCF listed is the highest BCF value reported in the cited reference.

<sup>b</sup>Calculated quantitative structure-activity relationship (QSAR) value as reported in ASTER.

BCF = bioconcentration factor; F = flow-through exposure system; S = static system; R = renewal system

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researchers still recommend caution in consuming some aquatic species (EPA 1993c; Keizer et al. 1991). This is in large measure because the mechanisms that fish and invertebrates use to metabolize diazinon are poorly understood and seem to vary widely from species to species. In addition, diazinon and its metabolites have not been widely monitored in aquatic species. Since some of the metabolites of diazinon are themselves toxic, a measure of caution may still be in order in cases where there is reason to believe edible fish or shellfish have had recent exposure to diazinon (Keizer et al. 1991). This is partially the basis for the EPA recommendation to states to consider routine monitoring for diazinon in edible fish and shellfish species as part of their state toxics monitoring programs particularly in those watersheds where extensive use of diazinon is identified (EPA 1993c).

Diazinon released in soil from its registered uses partitions to the atmosphere through volatilization, to surface water via runoff, and to groundwater as a result of leaching. According to Kenaga (1980), chemical compounds with a  $K_{oc}$  of  $<100$  are considered moderately to highly mobile; diazinon with a  $K_{oc}$  value of 40–432 (mean of 191), therefore, would be considered moderately mobile. Additional parameters influencing the leaching potential of this chemical include the soil type (e.g., clay versus sand), the amount of rainfall, the depth of the groundwater, and the extent of degradation. In laboratory tests of sand and organic soil, Sharom et al. (1980a) found that 26, 22, 11, 11, and 7% of the diazinon leached from sand (after five successive 200 mL rinses), respectively. A total of 95% of the diazinon added to the sand leached after 10 successive 200 mL rinses. In organic soil, however, only 3, 4, 11, 9, and 7% of the diazinon leached from soil (after five successive 200 mL rinses), respectively. Only 50% of diazinon added to the organic soil leached after 10 successive 200 mL rinses. While diazinon can show sorption in soils with high organic content ( $>3\%$ ), in most other soil types, diazinon has properties suggesting a moderate potential for leaching into groundwater (Arienzo et al. 1994; Sharom et al. 1980a). Arienzo et al. (1994) tested the adsorption and mobility of diazinon in 25 soils with different physicochemical properties. Diazinon was found to be slightly mobile in 80% of the soils tested (those with organic matter content  $<3\%$ ), and immobile in 20% of the soils tested (those with organic matter content  $>3\%$ ). The compound leached primarily from light soils with low organic matter content. Levanon et al. (1994) assessed the impact of plow tillage on microbial activity and the fate of diazinon and other pesticides in the top 5 cm soil layer. A higher leaching rate for diazinon was detected in plow tillage soils than in no-tillage soils after incubation for 21 days. The no-tillage soils were characterized by a higher organic matter content and higher microbial populations and activity than the plow tillage soils.

Arienzo et al. (1993) conducted a study of adsorption and mobility of diazinon in soils from aqueous media and mixtures of methanol-water and hexane-water. Adsorption of diazinon by soils from aqueous

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systems was related to organic matter content (i.e., the higher the organic content, the greater the adsorption). In methanol-water and hexane-water systems, the adsorption of diazinon by soils decreased. This situation may arise at hazardous waste disposal sites where pesticide waste residues and cosolvents may be encountered together. The presence of these organic solvents will increase the mobility (leachability) of diazinon in the soil and increase the potential for groundwater contamination. Diazinon has been detected in groundwater in the United States (Cohen 1986; EPA 1989; HazDat 2008), and in the Great Lakes region of Ontario, Canada (Frank et al. 1987, 1990b).

### 6.3.2 Transformation and Degradation

Diazinon is subject to a variety of abiotic and biotic degradation processes in all environmental compartments.

#### 6.3.2.1 Air

Diazinon, once released to the atmosphere, may be subject to direct photolysis since it absorbs light in the spectra above 290 nm (Gore et al. 1971). Glotfelty et al. (1990a), Schomburg et al. (1991), Seiber et al. (1993), and Zabik and Seiber (1993), all reported the presence of diazinon and its activated product (diazoxon) in atmospheric samples. Glotfelty et al. (1990a) believe that diazoxon is formed by atmospheric oxidation especially during the daylight hours. Schomburg et al. (1991) reported that diazinon undergoes transformation to diazoxon during atmospheric transport from agricultural to non-agricultural areas. Seiber et al. (1993) reported mean concentrations of diazinon of 76.8 ng/m<sup>3</sup> and of diazoxon of 10.8 ng/m<sup>3</sup> in air samples collected near fruit and nut orchards in Parlier, California. The half-life (first-order kinetics) for the vapor phase reaction of diazinon with hydroxyl radicals in the atmosphere is estimated to be 4 hours, assuming an atmosphere containing 5x10<sup>5</sup> hydroxyl radicals/m<sup>3</sup> at 25 EC (Meylan and Howard 1993).

#### 6.3.2.2 Water

Diazinon released to water may be subject to both abiotic degradation (i.e., hydrolysis and photolysis) and biotic degradation by microorganisms. The rate of abiotic degradation is influenced strongly by pH and temperature. In a laboratory study, Chapman and Cole (1982) reported that pH alone influenced the half-life of diazinon maintained in sterile water-ethanol (99:1) phosphate buffer solutions at 25 EC.

Degradation of diazinon was most rapid under acidic conditions with half-life values in weeks (days shown in parentheses) (first-order kinetics) of 0.45 (3.15), 2.0 (14), 7.8 (54.6), 10.0 (70), and 7.7 (53.9) at

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pH values of 4.5, 5.0, 6.0, 7.0, and 8.0, respectively. Garcia-Repetto et al. (1994) also studied the influence of pH on the degradation of diazinon in water-ethanol (9:1) solutions maintained between 15 and 31 EC. These authors reported estimated half-life values (first-order kinetics) for diazinon of 1.31, 8.57, and 8.19 days at pH values of 2, 7.5, and 8.7, respectively. The higher temperatures and lower pH conditions of this study may account for the more rapid degradation rates. Frank et al. (1991b) followed the degradation of diazinon in natural surface/groundwater samples at pH 8.2 that were either stored in the laboratory at 4 EC in the dark or at 21 EC under ambient indoor fluorescent light conditions for 125 days. Under the two temperature and light regimes the half-life values (first-order kinetics) of diazinon were 14 days (light at 21 EC) and 45 days (dark at 4 EC). Degradation was more affected by temperature, suggesting that hydrolysis was the primary mode of degradation (Frank et al. 1991b).

EPA (1976) reported that diazinon absorbs sunlight less than some of its organophosphate relatives, but that diazinon undergoes direct photolysis in water. The estimated half-life (first-order kinetics) for photolysis in aqueous solutions maintained in glass cells and irradiated with a mercury vapor lamp (>290 nm) was 1,000 hours (42 days). Frank et al. (1991b) investigated the degradation of diazinon in surface water and groundwater samples, but found little difference in the rate of diazinon degradation in light and dark conditions. The half-life (first-order kinetics) of diazinon of 88 days (light) and 99 days (dark) suggests that photolysis was not a major factor in degradation.

Scheunert et al. (1993) studied the effects of photodegradation (via exposure to sunlight) on diazinon dissolved in distilled water, in a humic acid aqueous solution, and in natural water samples from the Isar and Rhine Rivers and Lake Ammersee in Germany with comparable samples maintained in the dark at 25 EC. In the dark, river water had a higher diazinon degradation capacity than distilled water. The degradation capacity of natural water samples was further enhanced by exposure to sunlight. The highest degradation capacity was observed for the Rhine River water which also had the highest oxygen and hydroxyl ion concentration and the highest pH value (8.1) of the natural waters tested.

Mansour et al. (1997) studied the photodegradation of diazinon under UV irradiation in a water/soil suspension and found that diazinon was converted, among other products, to the isomeric isodiazinon. Photodegradation was increased in the presence of titanium dioxide, hydrogen peroxide or ozone, or by using natural-river or lake water.

Sharom et al. (1980b) studied the degradation of diazinon under laboratory conditions using both distilled water and natural water samples. Degradation was more rapid in natural water (pH 7.7) (12 weeks) than

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in sterilized natural water, sterilized distilled water, or distilled water (>16 weeks), suggesting that biodegradation of diazinon was occurring. Ferrando et al. (1992) conducted a laboratory microcosm study using both natural surface water and tap water. These experiments were conducted in aerated aquaria, maintained at 22 EC with a 12-hour light:dark period. The pH of the natural water was 9.0 and that of the tap water was 7.5. The half-life values (first-order kinetics) of 71 and 79 hours for the natural and tap water samples, respectively, both indicate rapid degradation. Under these experimental microcosm conditions, hydrolysis, photolysis, and biodegradation may all be operative in the natural water system. Wide discrepancies in the rates of diazinon degradation in water reported in the literature appear to be influenced by both abiotic and biotic factors.

Bondarenko et al. (2004) investigated the persistence of diazinon in natural waters from different locations within the Upper Newport Bay-San Diego Creek watershed located in central Orange County, California. First-order half-lives for diazinon were 6.3–14.0 and 25.0–28.3 days in natural water at 21 and 10 °C, respectively, and 51.1–54.9 days in sterilized water at 21 °C. The first-order half-lives for diazinon in seawater were 41.0 and 124.0 days at 21 and 10 °C, respectively. The results of the study suggest that under similar pH conditions, the persistence of diazinon may be prolonged in seawater. Sterilization greatly increased persistence of diazinon in freshwater, indicating that degradation in freshwater was largely attributed to microbial activity. Diazinon was found to be degraded primarily by abiotic processes in seawater, and the lack of microbial degradation likely contributed to its prolonged persistence in the seawater. Degradation in freshwater also showed temperature dependence, with significantly faster dissipation of diazinon at 21 °C when compared to 10 °C (Bondarenko et al. 2004).

Although diazinon has been detected in groundwater samples in both the United States and Canada (Cohen 1986; EPA 1989; Frank et al. 1987, 1990b; HazDat 2008), no studies were identified concerning diazinon transformation and degradation processes within aquifers. Based on theoretical considerations, abiotic hydrolysis mechanisms would be expected to degrade diazinon within a few months (Chapman and Cole 1982; Cowart et al. 1971).

### 6.3.2.3 Sediment and Soil

Once released to soils and sediments, diazinon can be degraded by hydrolysis, photolysis, and biodegradation by several genera of microorganisms. Microbial degradation appears to be the major pathway for the degradation of diazinon in soils; however, under anaerobic conditions, abiotic hydrolysis

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appears to be the most probable mechanism responsible for degradation of the compound under acidic soil conditions (Larkin and Tjeerdema 2000).

The influence of soil pH on the persistence of diazinon was studied by Chapman and Cole (1982). Diazinon degradation was found to be more rapid in organic soils with pH values of 6.1 and 5.2 than in mineral soils with pH values of 6.8 and 8.0, and was slightly more rapid in the more acidic organic soil. Schoen and Winterlin (1987) conducted an extensive study of the effects of various soil factors and organic amendments on degradation of diazinon. The factors affecting the rate of diazinon degradation in soil were pH, soil type, organic amendments, soil moisture, and pesticide concentration. Soil pH was a major factor affecting degradation. At a soil concentration of 100 ppm diazinon and 50% water saturation, estimated half-life values (first-order kinetics) at pH 4, 7, and 10 were 66, 209, and 153 days, respectively, in sandy loam; 49, 124, and 90 days, respectively, in clay loam; and 14, 45, and 64 days, respectively, in sandy loam amended with peat. Loss of diazinon occurred in the order of sandy loam with peat > clay loam > sandy loam. Addition of acidic peat to the soil lowered the pH and could have been responsible for increased hydrolysis. Degradation of diazinon in soil was most favorable when the pesticide was present at low concentrations in moist soil, amended with peat or acidified to a pH of 4, and least favorable at high diazinon concentrations in neutral or basic mineral soil (Schoen and Winterlin 1987). It has been observed that dissipation of diazinon from soil slows significantly as concentration increases above a certain level, possibly due to microbial toxicity which inhibits degradation (Felost et al. 2003). Prolonged persistence of diazinon in soil increases the potential for runoff and leaching.

In six types of soils, Somasundaram et al. (1991) reported that diazinon was hydrolyzed to 2-isopropyl-6-methyl-4-hydroxypyrimidine (IMHP) and that the degradation product was significantly more mobile in these soils than its parent compound diazinon. In an earlier study, Somasundaram et al. (1989) found that prior applications of IMHP did not enhance degradation of diazinon.

In a study of degradation of diazinon in three submerged tropical soils, only 2–6% of the originally applied diazinon remained 50–70 days post-application (Sethunathan and MacRae 1969). Degradation of diazinon was more rapid in nonsterilized soils, indicating microbial participation in two of the three soil types. In the third type (an acid clay soil), diazinon degradation was more rapid in the sterilized samples at pH 4.7, apparently because of the compound's instability under acid conditions. *Streptomyces* sp. isolated from the submerged soils could degrade the diazinon. In a field study of a treated cranberry bog by Szeto et al. (1990), disappearance of diazinon from irrigation ditch sediment (pH 4.4) and from sediment in an adjacent reservoir (pH 5.0) was equally rapid. These authors found that <1% of diazinon

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remained 38 and 22 days post-application in the irrigation ditch and reservoir sediments, respectively. In nonsterilized soil, diazinon degradation was faster at 100% water saturation than at 50% water saturation. These results suggest that microbial activity under anaerobic conditions plays an important role in diazinon degradation (Schoen and Winterlin 1987).

Photolysis of diazinon on soil surfaces was studied by Burkhard and Guth (1979). The effectiveness of photolysis in 24 hours was only slightly greater on moist soil surfaces (51%) than it was on dry soil surfaces (44%) at 45 EC. The major photolytic product identified for diazinon was 2-isopropyl-6-methyl-4-hydroxypyrimidine. This same reaction product was found for acid hydrolysis and photolysis in aqueous solutions or on soil.

Photochemical reactions in soil are significant only at the surface. When the surface soil is moist, photolysis may occur both with the portion dissolved in water and with the portion adsorbed to the soil. Mansour et al. (1997) studied the photochemical reaction of diazinon in water/soil suspensions in order to include the possible catalytic effects of adsorption. Under UV irradiation in a water/soil suspension, diazinon was converted to, among other products, the isomeric isodiazinon. Photodegradation was increased in the presence of titanium dioxide, hydrogen peroxide, or ozone, or by using natural river water or lake water.

Gunner and Zuckerman (1968) reported synergistic microbial degradation of diazinon by two microorganisms, *Arthrobacter* sp. and *Streptomyces* sp. When *Arthrobacter* sp. and *Streptomyces* sp. were incubated separately on growth media where diazinon was the primary carbon source, neither was able to convert the pyrimidinyl carbon to carbon dioxide. When incubated together, only 6% of the parent diazinon remained, and 94% was converted to two unidentified metabolites. Two microorganisms isolated from flood soils also were found to hydrolyze diazinon (Adhya et al. 1981). Diazinon was rapidly hydrolyzed within 24 hours by both *Flavobacterium* sp. and *Pseudomonas* sp. A hydrolysis product of diazinon, IMHP, was metabolized more rapidly by the *Flavobacterium* sp. than the *Pseudomonas* sp. More recently, oxypyrimidine was reported to be the major soil degradation product of diazinon and is considered to be more persistent than diazinon (Larkin and Tjeerdema 2000). Barik and Munnecke (1982) reported that an enzyme (parathion hydrolase) obtained from *Pseudomonas* sp. cultures could hydrolyze diazinon in soils. More than 98% of 10,000 ppm of diazinon in soil can be degraded within 24 hours if sufficient buffer and enzyme are added to the contaminated soil. The authors report that it is technically feasible to use parathion hydrolase to clean up diazinon spills in the environment.



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Levanon et al. (1994) studied the effects of plow tillage on microbial activity and the degradation of diazinon in the 0–5 cm soil layer. In no-tillage soils, higher microbial populations and activity were associated with higher mineralization rates of diazinon (45% mineralization after 76 days). Enhanced transformation rates played a role in minimizing leaching from no-tillage soils. Synergistic effects between fungi and bacteria in the degradation of diazinon were also observed. The authors noted that almost no mineralization of the compound occurred when either fungi or bacteria were selectively inhibited, demonstrating synergism between the two microbial communities. A higher proportion of diazinon leached from the plow tillage soils than from the no-tillage soils. Microbial population and activity measured as biomass, bacterial counts, hyphal length of fungi, and carbon dioxide evolution were all higher in samples of no-tillage soils.

**6.3.2.4 Other Media**

Michel et al. (1997) studied the fate of diazinon during the composting of leaves and grass. The yard trimmings were amended with  $^{14}\text{C}$  labeled diazinon and composted for 54 days. During composting, 11% of the  $^{14}\text{C}$ -diazinon was mineralized to carbon dioxide. A water extract of the finished compost contained 36% of the added  $^{14}\text{C}$  and analysis of this extract indicated that the  $^{14}\text{C}$  was in the form of 2-isopropyl-IMHP, a hydrolysis product of diazinon. The remaining fraction of  $^{14}\text{C}$  was unidentifiable or associated with a high molecular weight extract fraction. The results show that during the composting, a relatively small amount of diazinon is mineralized to carbon dioxide, while a majority is hydrolyzed to potentially leachable but less toxic IMHP, high molecular weight residues, and unextractable residues that are presumed to have low bioavailability.

Diazinon has been detected at a variety of waste water treatment plants. Zhang and Pehkonen (1999) investigated the oxidation of diazinon by aqueous chlorine and reported that the half-life is only several minutes at typical chlorine concentrations found in waste water treatment plants. The oxidation product of diazinon was reported to be diazoxon.

**6.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT**

Reliable evaluation of the potential for human exposure to diazinon depends in part on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of diazinon 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 diazinon levels monitored or estimated in the environment, it should also be noted that the amount of chemical identified analytically is not necessarily

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equivalent to the amount that is bioavailable. The analytical methods available for monitoring diazinon in a variety of environmental media are detailed in Chapter 7.

Most information on diazinon concentrations in various environmental media derived from large scale monitoring networks dates from before the mid-1980s and no longer reflects current conditions. There is a noticeable lack of national monitoring studies that would allow meaningful estimation of current diazinon concentrations associated with various environmental media. Reliable evaluation of the potential for human exposure to diazinon depends in part on the reliability of supporting analytical data from environmental samples and biological specimens.

#### 6.4.1 Air

Diazinon concentrations in the atmosphere were monitored in several national studies during the 1970s and 1980s and more recently in several regional studies. Diazinon has been measured in outdoor air samples in both rural and urban environments, near production facilities, and in indoor air (associated with its use for pest control in domestic and commercial buildings).

In a study of pesticide residues in ambient air sampled in 14–16 states during 1970, 1971, and 1972, diazinon was detected in 50% of the 2,479 samples analyzed, with a mean concentration of 2.5 ng/m<sup>3</sup> and a maximum concentration of 62.2 ng/m<sup>3</sup> (Kutz et al. 1976). Carey and Kutz (1985) reported that ambient air concentrations of diazinon collected from February through September 1980 in Perkin, Illinois, ranged from 1.3 to 10 ng/m<sup>3</sup>.

In a study of pesticide levels in ambient suburban air, diazinon was detected in 80, 80, and 40% of samples collected in three cities (Miami, Florida; Jackson, Mississippi; and Fort Collins, Colorado), respectively. The maximum diazinon concentration detected in each city was 3.9, 2.0, and 2.2 ng/m<sup>3</sup> for Miami, Florida; Jackson, Mississippi; and Fort Collins, Colorado, respectively (Kutz et al. 1976). During 1973–1974, diazinon concentrations in air were measured in urban Miami, Florida, and in the adjacent Everglades National Park. Urban diazinon levels ranged from not detectable to 3.3 ng/m<sup>3</sup> (1.5 ng/m<sup>3</sup> mean); corresponding levels in Everglades National Park ranged from not detectable to 1.9 ng/m<sup>3</sup> (0.6 ng/m<sup>3</sup> mean) (Lewis and Lee 1976). Nationwide, diazinon was detected in 48% of 123 urban air samples collected in 10 U.S. cities during 1980. The maximum diazinon concentration reported was 23 ng/m<sup>3</sup> (mean 2.1 ng/m<sup>3</sup>) (Carey and Kutz 1985).

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Most recently, non-occupational exposure to diazinon among residents of two U.S. cities (Jacksonville, Florida, and Springfield, Massachusetts) was studied over three seasons: summer 1986, spring 1987, and winter 1988 (Whitmore et al. 1994). The study focused primarily on inhalation exposures with primary environmental monitoring consisting of 24-hour indoor and outdoor air. For the Jacksonville, Florida, population, the estimated mean diazinon concentrations were 85.7–420.7 ng/m<sup>3</sup> in indoor air and 1.1–13.8 ng/m<sup>3</sup> in outdoor air. For the Springfield, Massachusetts, population, mean exposures were much less. The estimated diazinon concentrations were 2.5–48.4 ng/m<sup>3</sup> in indoor air and 8.2–9.2 ng/m<sup>3</sup> in outdoor air.

Ambient diazinon concentrations were measured under foggy atmospheric conditions in and around the Central Valley of California (Parlier, California), which is a prime agricultural area dominated by fruit, nut, and citrus orchards (Glotfelty et al. 1990a; Seiber et al. 1993; Zabik and Seiber 1993). In fog, diazinon concentration was 1.6 ng/m<sup>3</sup> and diazoxon (the oxon transformation product) concentration was 0.82 ng/m<sup>3</sup>. In a similar study, Schomburg et al. (1991) analyzed air and fog near Monterey, California, to determine whether the uptake of diazinon in advected oceanic fog was different from uptake in fog collected under stagnant inversion conditions in the Central Valley of California. Fog water concentrations of diazinon ranged from 0.15 to 4.8 µg/L in coastal areas; higher concentrations ranging from 0.31 to 18 µg/L were found in the Central Valley area. Diazinon and diazoxon favored the aqueous phase in foggy atmosphere, with 62.4 and 87.8%, respectively, reported in the aqueous phase. Zabik and Seiber (1993) studied the atmospheric transport of diazinon from California's Central Valley to the Sierra Nevada Mountains. Air samples collected from January through February 1991 represented the simultaneous collection of both vapor and particulate phases. Concentrations of diazinon and its oxon were 13–10,000 pg/m<sup>3</sup> (0.013–10 ng/m<sup>3</sup>) and 4–3,000 pg/m<sup>3</sup> (0.004–3 ng/m<sup>3</sup>), respectively, for samples collected at the 114 m elevation and 1.4–12 pg/m<sup>3</sup> (0.0014–0.012 ng/m<sup>3</sup>) and 1.8–13 pg/m<sup>3</sup> (0.0018–0.013 ng/m<sup>3</sup>), respectively, at the 533 m elevation. The pesticide concentrations in air samples decreased with distance and elevation moving east from the Central Valley into the higher elevations of the Sierra Nevada Mountains. At times, air concentrations of diazinon at the 114 m elevation were 1,000 times greater than concentrations detected at 533 m elevation. Concentrations at the 1,920 m elevation were typically below the limit of quantification. Wet deposition samples (rain and snow) collected at the 114 m elevation contained up to 6,100 pg/mL diazinon and 2,300 pg/mL diazoxon. Diazinon was detected in 100% of air samples collected from over the Mississippi River from New Orleans, Louisiana, to St. Paul, Minnesota, during the first 10 days of June 1994 at a mean concentration of 0.08 ng/m<sup>3</sup> (Majewski et al. 1998). The maximum concentration detected, 0.36 ng/m<sup>3</sup>, was detected near a major metropolitan area.

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Diazinon residues in ambient air sampled within 800 m of two pesticide formulation plants in Arkansas (from 1970 to 1972) and within 275 m of a pesticide formulation plant in Tennessee (in 1971) were 0.3–18.0 ng/m<sup>3</sup> (mean 2.2 ng/m<sup>3</sup>) and 0.5–27.9 ng/m<sup>3</sup> (mean 7.3 ng/m<sup>3</sup>), respectively (Lewis and Lee 1976).

In addition to its presence in the ambient atmosphere, diazinon also has been monitored in both outdoor and indoor air associated with its use in a variety of domestic, commercial, and occupational exposure situations. Exposure to diazinon from its use in lawn and home garden applications was evaluated by Davis et al. (1983). Diazinon was mixed with water and sprayed using compressed air sprayers or hose-end sprayers, and potential respiratory and dermal exposures were estimated from residues collected from respirator filters, body pads, and hand rinsings. These authors reported mean respiratory exposures of 1.9, 2.9, and 7.4 µg/hour associated with use of compressed air sprayers on lawns, compressed air sprayers on shrubs, and hose-end sprayers on lawns, respectively. The amount of diazinon collected in the respiratory pads was negligible compared to the amount collected on dermal pads. Total dermal exposures were 5,700, 7,500, and 29,000 µg/hour, respectively, for the three sprayer types; however, dermal exposure of the hands alone accounted for ≥85% of the total dermal exposure for each sprayer type.

Diazinon air concentrations related to vapors released from pest control strips were measured by Jackson and Lewis (1981). Diazinon levels in indoor air increased from 0.32 µg/m<sup>3</sup> at 6 hours after application of the pest strips to 1.34 µg/m<sup>3</sup> on day 15, and then declined to 1.21 µg/m<sup>3</sup> on day 30. Air sampling in a retail garden store where pesticide containers with diazinon were displayed showed an average diazinon concentration of 3.4 µg/m<sup>3</sup> (Wachs et al. 1983).

Currie et al. (1990) evaluated the concentrations of diazinon in indoor air and on working surfaces for a period of 10 days after application in commercial offices. The highest concentrations of diazinon (163 and 158 µg/m<sup>3</sup>) were measured 4 hours post-application in two empty offices, while the concentration in the furnished office was 27 µg/m<sup>3</sup>. One day post-application, levels were 125 and 70 µg/m<sup>3</sup> in the two empty offices, and 27 µg/m<sup>3</sup> in the furnished office. Air concentrations of diazinon continued to decline and on day 6 post-application were approximately 35 µg/m<sup>3</sup> in the empty offices and 8 µg/m<sup>3</sup> in the furnished office. Airborne levels of diazinon were distinctly lower in the furnished office, and this was attributed to obstruction of the applicator's spraying path by office furniture so that a lower amount of diazinon was applied. Diazinon deposition on aluminum plates was measured as an indicator of surface contamination, measurements ranged from 0.4–15 ng/cm<sup>2</sup>. No overall decrease in surface contamination occurred over time. Plates suspended 1.5–2.1 m above the floor generally exhibited higher

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diazinon levels 24 hours post-treatment than at 1–2 hours post-treatment. The authors believe that this was a result of evaporation of diazinon from the carpeted floor augmented by air turbulence. Diazinon contamination measured by surface wipes on furniture and foil on carpet ranged from 13 to 38 ng/cm<sup>2</sup>.

Diazinon levels in indoor air were monitored in an animal facility treated monthly with a 1% aqueous diazinon formulation (Williams et al. 1987). Indoor air sampling was conducted in two areas frequented regularly by facility personnel, the lounge and cage-washing areas. The lounge areas were enclosed rooms while the cage-washing areas were open-ended and were in effect part of the corridor system of the facility. Air samples were collected using adsorbent sampling tubes (Supelco-20 P) for 4 hours at 1.8 L/minute just prior to spraying on days 0, 28, and 56, approximately 16–20 hours post treatment, and at various intervals thereafter. Diazinon levels increased immediately after spraying, but decreased rapidly to 2–3 µg/m<sup>3</sup> in <1 day and continued to decrease to <0.05 µg/m<sup>3</sup> until the next spraying. During many months of diazinon application there was little buildup in background diazinon air levels (<0.5 µg/m<sup>3</sup>).

Lenhart and Kawamoto (1994) reported air concentrations of up to 297 µg/m<sup>3</sup> in greenhouse air after spray applications of an emulsifiable concentrate of diazinon, under the trade name Clean Crop AG500, and concentrations up to 3,030 µg/m<sup>3</sup> in greenhouse air after a 4-hour cold fogging application of the same formulation.

Palmgren and Lee (1984) collected samples of grain dust (dust accumulated in the dust collection systems of grain elevators) from six grain elevators located in the New Orleans, Louisiana, area to evaluate potential occupational exposures of grain elevator personnel. Diazinon concentrations in grain dust were <0.01 µg/g for all 31 samples collected.

#### 6.4.2 Water

Carey and Kutz (1985) reported that the maximum diazinon residue collected in a national surface water monitoring program conducted from 1976 to 1980 was 2.38 µg/L and that diazinon was detected in only 1.2% of the samples collected. Pereira and Hostettler (1993) conducted a study of the Mississippi River and its tributaries during 1991 and 1992. These authors reported that diazinon was detected in water samples from the Illinois River at concentrations of 20 ng/L and from several sites on the mainstem of the lower Mississippi River at concentrations ranging from 4 to 10 ng/L. During 1991, Domagalski and Kuivila (1993) monitored diazinon concentrations in water and suspended sediment collected at various

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sites in San Francisco Bay during low river discharge and after spring rain events. Diazinon was detected in water only after the spring rains and most (98%) of the diazinon was in the dissolved phase. Concentrations dissolved in the water column ranged from 4.6 to 14.6 ng/L. The authors suggest that diazinon may be close to equilibrium with respect to sorption or desorption on suspended sediment particles.

In the Great Lakes region, diazinon was detected in surface waters in several river basins in southern Ontario, Canada. Braun and Frank (1980) monitored surface water concentrations of 8 organochlorine and 12 organophosphate pesticides in 11 agricultural watersheds in southern Ontario. All watersheds drained into the Great Lakes. Diazinon residues as a result of field use were detected in only one watershed, but the chemical was repeatedly detected in 34% of samples (1975–1976) and 74% of samples (1976–1977) collected from one creek. The source of the diazinon was traced to its indoor use to control flies in a series of mushroom houses that discharged via a drainage tile system directly to the creek. The maximum residues of diazinon in the stream were 140 µg/L (5.75 µg/L mean) and 26 µg/L (1.02 µg/L mean) in 1975–1976 and 1976–1977, respectively. Frank and Logan (1988) measured pesticide and industrial chemical residues at the mouth of the Grand, Saugeen, and Thames Rivers in southern Ontario, Canada, from 1981 through 1985. River water samples collected at the mouths of the three rivers (that drain into the Great Lakes) were analyzed for 20 herbicides, 3 fungicides, and 25 insecticides including diazinon. One water sample collected during May through August 1982 contained a mean diazinon concentration of 0.21 µg/L. Maguire and Tkacz (1993) monitored concentration of pesticides in surface water near the mouths of the Yamaska River in Quebec, Canada, and five of its tributaries during 1986 and 1987. Diazinon was detected at the mouth of the Yamaska River at concentrations ranging from 2.1 to 11.9 ng/L, at the mouth of the Saint-Nazaire River at concentrations ranging from 3.1 to 26.7 ng/L, and at the mouth of the Salvail River at concentrations ranging from 1.1 to 4.9 ng/L. Frank et al. (1990a) conducted a survey of 211 rural ponds in southern Ontario and measured concentrations of 29 herbicides, fungicides, and insecticides including diazinon. Two ponds were found to be contaminated with diazinon, and residues in pond water ranged from 0.6 to 1.7 µg/L (1.2 µg/L mean). The source of the diazinon in these two cases was attributed to accidental pesticide spills during agricultural application. In the U.S. Geological Survey's National Water-Quality Assessment Program 1999–2000, surface water samples from the Yakima River Basin, Washington were collected at 34 sites and analyzed for the occurrence of pesticides (USGS 2002a). Diazinon was detected in 16 of 98 surface water samples at concentrations ranging from <0.002 to 0.169 µg/L. Banks et al. (2005) collected 1,243 surface water samples at 70 monitoring stations from rural and urban streams in Denton, Texas during the years of 2001–2004 and monitored for diazinon before and after the EPA ban on its residential uses. The total

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number of samples having diazinon concentrations above the lower limits of detection significantly decreased from 2001 to 2004. The maximum and average concentrations, respectively, of diazinon detected each year were 2.58 and 0.32 µg/L in 2001, 1.67 and 0.10 µg/L in 2002, 1.91 and 0.06 µg/L in 2003, and 0.85 and 0.04 µg/L in 2004. These results indicate that the phasing out of residential uses of diazinon has led to a significant decrease in surface water occurrences.

Diazinon concentrations in water have also been monitored in the United States and in several Canadian studies associated with the use of the compound in agricultural applications. Kendall et al. (1993) monitored diazinon residues in ponds and creeks adjacent to a golf course in coastal Washington where two turf applications of diazinon were made at a rate of 2.2 kg active ingredient per hectare. A maximum diazinon residue of 17 µg/L was measured in the study area ponds and creeks. Wan et al. (1994) monitored concentrations of diazinon and six other organophosphate pesticides in farm ditches of the lower Fraser River Valley of British Columbia, Canada, from July to December 1991. These authors reported that diazinon was consistently found in ditch water (81% of samples) at 7 locations with a mean concentration of 0.07 µg/L (range of 0.01–0.34 µg/L). The percentage of positive detections for diazinon in water samples was 81%. The presence of diazinon in ditch water was correlated with consistent detection of diazinon residues in soils from nearby fields. Szeto et al. (1990) monitored the persistence of diazinon in coastal cranberry bogs and adjacent surface waters in British Columbia, Canada. Bogs were treated with two applications of diazinon 5G (granules) at a rate of 6 kg active ingredient per hectare approximately 2 weeks apart. One day after the first and second applications, maximum concentrations of diazinon in water in an irrigation ditch were 338 and 456 µg/L, respectively. Maximum concentrations in an adjacent reservoir were 78.5 and 58.1 µg/L for the first and second treatments, respectively. Water samples collected immediately outside the diked bog area contained a maximum of 29.1 µg/L diazinon, but concentrations were usually <10 µg/L. Tributary water 100 m downstream from the cranberry bog site contained a maximum diazinon residue of 2.8 µg/L. Li et al. (2002) monitored diazinon levels in water samples collected in June, August, and October 2000 from 13 agricultural water ditches in British Columbia, Canada near fields where diazinon was applied. Diazinon was found to range from not detected (<4 ng/L) to 259 ng/L, indicating that after application, high concentrations can reach local waterways. High diazinon concentrations, such as the 259 ng/L detection, may be of ecological relevance given that concentrations of 350 ng/L have been associated with toxic effects in aquatic organisms. Between March 1993 and May 1994, Gruber and Munn (1998) measured median and maximum diazinon concentrations of <0.002 and 0.018 µg/L, respectively, in Crab Creek Lateral, in which most of the flow is excess irrigation water, located in the Central Columbia Plateau in central Washington state. EPA's Water Resources Assessment for diazinon provided estimated environmental concentrations of diazinon

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in surface water as a result of the highest label application rate on seven crop types calculated using the Pesticide Root Zone Model version 3.1 (PRZM) and Exposure Analysis Modeling System Version 2.97.5 (EXAMS) (EPA 1999). The peak and yearly average estimated environmental concentrations of diazinon ranged from 8.89 to 429 µg/L and from 1.61 to 58.7 µg/L, respectively.

Suzuki et al. (2003) monitored pesticide residues in rainwater from July 1999 to July 2000 at five sampling sites in Utsunomiya, eastern Japan. The annual deposited amount as an average value among the sampling sites for diazinon was 38.3 µg/m<sup>2</sup>. In the 82 rainwater samples collected over the year, the maximum and mean concentrations of diazinon detected were 0.324 and 0.079 µg/L, respectively.

Recently, acute toxicity of sewage treatment plant effluents to aquatic bioassay testing organisms in the United States has been tied to diazinon (Amato et al. 1992; Burkhard and Jenson 1993). Given the considerable use of diazinon in urban areas, diazinon in sewage treatment effluents is not unexpected. Urban nonpoint source inputs from diazinon-impregnated yard wastes, runoff from treated lawn and garden areas, or illegal dumping may require increased pollution prevention efforts through the National Pollution Discharge Elimination System (NPDES) program in many larger cities (Amato et al. 1992; Burkhard and Jenson 1993). A maximum diazinon residue of 1.7 µg/L in POTW effluents was associated with the toxic fraction in effluent bioassay tests with *Ceriodaphnia dubia* (Burkhard and Jenson 1993). Amato et al. (1992) suggest that the significance of detecting diazinon at acutely toxic concentrations in municipal waste water may indicate a more widespread problem.

Schiff and Sutula (2004) detected diazinon in 93% of 128 storm-water runoff samples from eight different land uses over five storm events in urban southern California watersheds. The mixed agricultural land use had the highest flow-weighted mean diazinon concentration of 4,076.0 ng/L, followed by commercial and nursery agricultural land use with mean diazinon concentrations of 324.0 and 148.0 ng/L, respectively. High and low-density residential land use had mean concentrations of 99.2 and 67.6 ng/L, respectively, industrial land use had 89.6 ng/L, recreational land use had 63.2 ng/L, and the open space site had the lowest concentration of diazinon of <20 ng/L. Diazinon was detected in 100% of the samples collected at the mass emission sites Ballona Creek and Santa Monica Canyon, located in Santa Monica Bay, with flow-weighted mean concentrations of 242.9 and 452.3 ng/L, respectively.

In a groundwater contamination study of 28 of California's 58 counties that evaluated over 50 pesticides (from both point and nonpoint sources), diazinon was detected in 12 samples (Cohen 1986). Diazinon is included as an analyte of interest in the EPA Pesticides in Ground Water Database (EPA 1989) and was



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detected at two sites. A detection in California was related to point source contamination (residue level was unspecified), and a detection of 478 µg/L (maximum) and 162 µg/L (mean) in Mississippi was in an area where appreciable agricultural use of pesticides occurs. In the Great Lakes region, diazinon was found in a survey of rural wells in southern Ontario, Canada, monitored between 1979 and 1984 (Frank et al. 1987) and in farm wells monitored between 1986 and 1987 (Frank et al. 1990b). However, no concentrations of diazinon in groundwater were provided by these authors. In the U.S. Geological Survey's National Water-Quality Assessment Program 1992–1996, 2,485 groundwater sites were sampled in 20 of the nation's major hydrologic basins were analyzed for 90 pesticide compounds (Kolpin et al. 2000). Diazinon was detected at a frequency of 1.30% at 2,459 sites with a maximum detected concentration of 0.16 µg/L.

Only limited data on the concentration of diazinon in drinking water are available since drinking water facilities are not required to monitor for diazinon (EPA 1999). EPA's Water Resources Assessment for diazinon provided a drinking water assessment and set the upper bound on the drinking water exposure estimate through modeling with surface and groundwater data (EPA 1999). The estimated diazinon acute exposures in drinking water were 2.3–22, 3.0–22, and 0.90 µg/L based on agricultural and non-agricultural use surface water and groundwater, respectively. The estimated diazinon chronic exposures in drinking water were 0.19–5.8, 0.46–5.8, and 0.90 µg/L based on agricultural and non-agricultural use surface water and groundwater, respectively. Eitzer and Chevalier (1999) detected diazinon in 5 of 53 residential drinking wells at an average concentration of 0.02 µg/L in a single town in south-central Connecticut which relies on groundwater for its potable water source. In the EPA's National Survey of Pesticides in Drinking Water, no diazinon was detected (limit of detection of 1.10 µg/L) in 1,349 wells (783 rural domestic wells and 566 community water system wells) randomly selected and sampled once for diazinon in 38 states (EPA 1999). In February 2005, the EPA's Office of Water added diazinon to the Drinking Water Contaminant Candidate List (EPA 2006j). EPA uses this list of unregulated contaminants to prioritize research and data collection efforts to help determine whether the specific contaminant should be regulated.

### 6.4.3 Sediment and Soil

Diazinon has not been the focus of many national soil or sediment monitoring programs in the United States, but has been monitored in regional studies associated with agricultural applications in both the United States and Canada. In a national surface water quality monitoring study (1976–1980), diazinon was detected in 0.5% of the sediment samples analyzed, with a maximum residue of 7.1 µg/L (Carey and

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Kutz 1985). Domagalski and Kuivila (1993) reported concentrations of diazinon in suspended sediments from various sites from San Francisco Bay ranging from not detected to 2.8 ng/g.

Soil contamination of diazinon ranging from 95.5 mg/m<sup>2</sup> (2 hours post-application) to 35.6 mg/m<sup>2</sup> (342 hours post-application) resulted from spray applications of 4.5 kg diazinon (50 WP formulation) per hectare to a dormant peach orchard in the Central Valley of California (Glotfelty et al. 1990b). Diazinon concentrations in sediments of a cranberry bog treated with two applications of diazinon (Diazinon 5G at 6 kg active ingredient per hectare) were measured by Szeto et al. (1990). These authors reported that the highest diazinon residues were 21 µg/g (wet weight) in sediments of irrigation ditches collected 4 days post-application. The maximum sediment concentration measured in an adjacent reservoir was 2 µg/g. Four days post-application, the maximum sediment concentration was 80 µg/kg in a waterway outside the diked bog and only 10 µg/kg in a tributary 100 meters downstream from the bog. Wan et al. (1994) monitored ditch water, soils, and sediments from July to December 1991 in an agricultural area in the lower Fraser River Valley of British Columbia, Canada. Diazinon concentrations in ditch sediment were detected at three sites; the mean concentrations were 8, 2, and 38 µg/kg at the Vancouver, Cloverdale, and Sumas Prairie sites, respectively. Diazinon was also detected in topsoil (<5 cm deep) at five sites; the mean concentrations were 268 µg/kg (range of 2–3,307 µg/kg), 5 µg/kg (range of 1–9 µg/kg), 769 µg/kg (range of 13–2,862 µg/kg), 13 µg/kg (range of 4–30 µg/kg), and 39 µg/kg (range of 1–236 µg/kg) at the Westham Island, Ladner, Burnaby, Cloverdale, and Sumas Prairie sites, respectively. The concentrations at all these stations declined from July to December. Diazinon concentrations ranging from 0.5 to 5.4 ng/g dry weight were measured in sediments collected from the Salton Sea, an agricultural drainage reservoir in California, in 2000 and 2001 (Sapozhnikova et al. 2004). Sediment concentrations were 64% higher in 2001 than in 2000.

Sediment samples collected from six sites in the Petaluma River, Sonoma Creek, and Napa River, which feed into the San Pablo Bay, California contained diazinon concentrations ranging from 8.45 to 13.10 µg/kg (Baum et al. 2001).

#### 6.4.4 Other Environmental Media

Braun and Frank (1980) reported diazinon residues in three fish species collected from a creek in southern Ontario, Canada, contaminated from a point source discharge. Tissue residues for the three edible fish species were 18 µg/g in the brown bullhead (*Ictalurus nebulosus*), 17 µg/g in the black crappie (*Pomoxis nigromaculatus*), and 92 µg/g in the gizzard shad (*Dorosoma cepedianum*). The maximum diazinon

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concentrations measured in the contaminated creek water for 1975–1976 and 1976–1977 were 140 µg/L (5.75 µg/L mean) and 26 µg/L (1.02 µg/L mean), respectively. Sapozhnikova et al. (2004) reported diazinon residues in tilapia (*Tilapia mossambique*) and orange mouth Corvina (*Cynoscion xanthulu*) collected from the Salton Sea, an agricultural drainage reservoir in California, in May 2001. In the corvina, diazinon was found in the muscle, liver, gonads, and gills at mean concentrations of 5.4, 17.2, 6.2, and 3.6 ng/g wet weight, respectively. In the tilapia, diazinon was found in the muscle, liver, gonads, and gills at mean concentrations of 4.4, 8.8, 5.2, and 2.4 ng/g wet weight, respectively.

Concentrations of diazinon in ready-to-eat foods were monitored for 10 years from 1982 to 1991 through the FDA's Revised Market Basket Survey (KAN-DO 1995). Diazinon was detected in 894 samples of 144 different foods at a mean concentration of 0.0019 µg/g. In the EPA's Revised Organophosphate Pesticides Cumulative Risk Assessment, a summary of residue monitoring data on organophosphate pesticides in food for the years 1994–2000 was reported (EPA 2002). The detection of diazinon in these various foods and its concentration are presented in Table 6-3. The report also included a summary of FDA Total Diet Study analyses on organophosphate pesticides on meats for the years 1991–1999. The mean concentrations of diazinon residues found in meats (in mg/kg) were 0.0009 in beef steak, loin, pan-cooked, 0.0008 in pork chop, pan-cooked, 0.009 in lamb chop, pan-cooked, and 0.002 in lamb chop, pan-cooked (sample 2). Residues of diazinon in levels of 0.005–0.586 mg/L have been reported in milk (Salas et al. 2003).

The frequency of occurrence of diazinon detections in the FDA Total Diet Study was 9% in 1989 (FDA 1990), 6% in 1990 (FDA 1991), 4% in 1991 (FDA 1992), 5% from 1991 to 1993 (FDA 1994), 5% in 1994 (FDA 1995), 3% in 1995 (FDA 1996), 2.4% in 1996 (FDA 1998), and 2% in 2003 (FDA 2005b). Diazinon intakes in µg/kg body weight/day, estimated for the total diet analyses were 0.0031, 0.0034, and 0.0017 in 1989 (FDA 1990); 0.0026, 0.0022, and 0.0017 in 1990 (FDA 1991); and 0.0049, 0.0022, and 0.0022 in 1991 (FDA 1992) for 6–11-month-old infants, 14–16-year-old-males, and 60–65-year-old-females, respectively.

## 6.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

While no quantitative information is available on the percentage of diazinon released to each environmental compartment, diazinon can be emitted to any or all environmental media (air, surface water, groundwater, and soil) depending on the source of the release, formulation used, and prevailing environmental conditions. In order to mitigate the exposure and risk to the general population, especially

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**Table 6-3. Diazinon Residues in Various Foods from 1994 to 2000**

Food item	Number analyzed	Number of detections	Average concentration (ppm) <sup>a</sup>	Maximum concentration (ppm)
Apple juice	1,554	0	0	0
Apples	2,472	19	1.2x10 <sup>-4</sup>	0.12
Apples (single serving)	377	1	1.9x10 <sup>-5</sup>	0.007
Bananas	1,126	0	0	0
Broccoli	634	0	0	0
Cantaloupe	1,640	6	4.7x10 <sup>-5</sup>	0.024
Carrots	2,071	79	6.8x10 <sup>-4</sup>	0.086
Celery	176	8	4.32x10 <sup>-4</sup>	0.027
Cherries	275	2	5.8x10 <sup>-5</sup>	0.013
Corn syrup	454	0	0	0
Cucumbers	1,467	8	1.51x10 <sup>-4</sup>	0.083
Grape juice	1,378	0	0	0
Grapes	2,625	38	3.04x10 <sup>-4</sup>	0.15
Green beans (canned)	854	0	0	0
Green beans (fresh)	1,898	5	6.18x10 <sup>-4</sup>	1.1
Green beans (frozen)	743	11	1.24x10 <sup>-4</sup>	0.019
Lettuce	1,616	47	3.82x10 <sup>-4</sup>	0.16
Milk	1,366	0	0	0
Nectarines	345	0	0	0
Orange juice	1,392	0	0	0
Oranges	2,636	0	0	0
Peaches (canned)	754	0	0	0
Peaches (fresh)	1,623	97	9.57x10 <sup>-4</sup>	0.16
Peaches (single serving)	534	29	0.0012	0.23
Peanut butter	716	0	0	0
Pears (canned)	737	2	4.9x10 <sup>-5</sup>	0.018
Pears (fresh)	1,779	39	3.56x10 <sup>-4</sup>	0.094
Pears (single serving)	696	6	2.28x10 <sup>-4</sup>	0.084
Pineapples	364	0	0	0
Potatoes	1,770	1	2x10 <sup>-6</sup>	0.003
Poultry (adipose tissue)	476	2	1.71x10 <sup>-4</sup>	0.04542
Poultry (liver)	479	1	2.4x10 <sup>-5</sup>	0.011676
Poultry (muscle)	145	1	1.34x10 <sup>-4</sup>	0.01944
Soybean grain	748	8	5.1x10 <sup>-5</sup>	0.01
Spinach (canned)	863	0	0	0
Spinach (fresh)	1,638	40	9.9x10 <sup>-4</sup>	0.39
Spinach (frozen)	715	8	1.17x10 <sup>-4</sup>	0.024

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**Table 6-3. Diazinon Residues in Various Foods from 1994 to 2000**

Food item	Number analyzed	Number of detections	Average concentration (ppm) <sup>a</sup>	Maximum concentration (ppm)
Strawberries (fresh)	1,768	16	1.26x10 <sup>-4</sup>	0.049
Strawberries (frozen)	155	0	0	0
Sweet bell peppers	1,468	20	1.51x10 <sup>-4</sup>	0.061
Sweet corn (canned)	652	0	0	0
Sweet corn (fresh)	19	0	0	0
Sweet corn (frozen)	635	0	0	0
Sweet peas (canned)	746	0	0	0
Sweet peas (fresh)	9	0	0	0
Sweet peas (frozen)	703	10	1.82x10 <sup>-4</sup>	0.049
Sweet potatoes	1,559	3	8x10 <sup>-6</sup>	0.007
Tomatoes (canned)	737	1	7x10 <sup>-6</sup>	0.005
Tomatoes (fresh)	1,962	12	9.1x10 <sup>-5</sup>	0.09
Wheat	1563	24	2.39x10 <sup>-4</sup>	0.028
Winter squash (fresh)	1,216	3	2.1x10 <sup>-5</sup>	0.015
Winter squash (frozen)	470	1	1.5x10 <sup>-5</sup>	0.007

<sup>a</sup>Nondetects were counted as zero in calculating the average.

Source: EPA 2002

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children, the EPA has phased out all residential uses of diazinon as of December 2004 (EPA 2004b). General population exposure to diazinon may occur through three routes: dermal contact, inhalation, and ingestion of contaminated food or drinking water. Past major routes of exposure to diazinon for the general population were through dermal contact directly with the chemical during domestic application for control of home and garden pests; through dermal contact with diazinon-treated plant materials such as grass clippings; or through dermal contact with treated surfaces (e.g., furniture) in domestic or office buildings. For children particularly, potential sources of exposure were related to the indoor application of diazinon on furniture, rugs, and flooring and from translocation from pets. The general population may also be exposed to diazinon through inhalation of contaminated ambient (outdoor) air particularly in agricultural areas where diazinon is extensively used or in urban areas where it is applied to lawns and gardens. Since many commercial buildings and residential buildings were sprayed with diazinon or used pest control strips that vaporize diazinon, there has been the possibility of exposure from inhalation of vapors in these diazinon-treated indoor air spaces. The oral route of exposure may include ingestion of foods contaminated with small residues of diazinon or consumption of contaminated drinking water.

Davis et al. (1983) reported that dermal exposure (rate of contact with skin) to diazinon from spray applications of the compound for home and garden applications ranged from 5,700 to 29,000  $\mu\text{g}/\text{hour}$  depending on the type of sprayer used. The mean respiratory exposures ranging from 1.9 to 7.4  $\mu\text{g}/\text{hour}$ , were negligible compared to the dermal exposures. In addition, these authors reported that dermal exposure of the hands, which accounted for 85% or more of the total dermal exposure, could be easily reduced by the use of protective gloves.

Pesticides may be transported into homes by translocation of vapors and by track-in from outdoors on shoes, clothing, and animal fur. Factors such as the activities of children and pets might serve as an important vehicle for transport of diazinon into homes. Morgan et al. (2001) conducted a 15-day study in 1999 to investigate the potential for a family with an indoor/outdoor pet dog to transport diazinon into a home and the potential exposure to the residents following lawn application. Entryway deposits on the doormats had diazinon residue levels as high as 135.0  $\mu\text{g}/\text{g}$  3 days after application. Airborne levels in the living room of the home were at least 50 times above background levels at 0, 3, 9, and 15 days after application, with the highest concentration being 0.18  $\mu\text{g}/\text{m}^3$  the day of application. The living room carpet contained diazinon residues six times greater than background levels at concentrations of 4.28, 3.59, and 3.56  $\mu\text{g}/\text{g}$  at 3, 9, and 15 days post-application. The dog had diazinon residues on its paws as high as 2.27  $\mu\text{g}$  the day of application and 1.39  $\mu\text{g}$  3 days post-application, and also had residues as high as 0.47  $\mu\text{g}/\text{g}$  on its fur. The data show that diazinon residues were being physically tracked in by the pet

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and humans and also redistributed into the indoor air. A gradient of diazinon residues was found from the soil to the entryway and into the living room of the home. Also, the dog is shown to be a good vehicle for the uptake, transfer, and translocation into the home and is likely to expose the occupants through direct contact, such as petting and playing (Morgan et al. 2001).

A study conducted by Lewis et al. (2001) also demonstrated the importance of translocation of pesticides from areas of application to surfaces accessible for human contact and the potential exposure through inhalation, dermal contact, and ingestion. Potential indoor air inhalation exposures to diazinon after indoor application were estimated to be as high as 0.5  $\mu\text{g}/\text{kg}/\text{day}$ . Multimedia sampling at a subset of homes in Arizona participating in EPA's National Human Exposure Assessment Survey was conducted in order to assess residential environmental exposure to pesticides (Gordon et al. 1999). Diazinon was found in 53% of the house dust samples at  $<0.02\text{--}50.5 \mu\text{g}/\text{m}^2$ ; indoor air, 63%,  $<0.002\text{--}20.5 \mu\text{g}/\text{m}^3$ ; hand wipes, 32%,  $<0.01\text{--}18.4 \mu\text{g}$ ; and foundation soil (2.5 cm depth), 37%,  $<0.007\text{--}7 \mu\text{g}/\text{g}$ .

Non-occupational exposure to diazinon for residents of two U.S. cities (Jacksonville, Florida, and Springfield, Massachusetts) was studied over three seasons: summer 1986, spring 1987, and winter 1988 (Whitmore et al. 1994). The study focused primarily on inhalation exposures with primary environmental monitoring consisting of 24-hour indoor air, personal air, and outdoor air. For the population of Jacksonville, Florida, the mean diazinon concentration ranges were  $85.7\text{--}420.7 \text{ ng}/\text{m}^3$  for indoor air,  $1.1\text{--}13.8 \text{ ng}/\text{m}^3$  for outdoor air, and  $89.0\text{--}321.6 \text{ ng}/\text{m}^3$  for personal air. For the population in Springfield, Massachusetts, mean exposures were much less. The diazinon concentrations were  $2.5\text{--}48.4 \text{ ng}/\text{m}^3$  for indoor air,  $8.2\text{--}9.2 \text{ ng}/\text{m}^3$  for outdoor air, and  $1.4\text{--}10.1 \text{ ng}/\text{m}^3$  for personal air. The mean air exposure for diazinon in Jacksonville, Florida, was 1,380  $\text{ng}/\text{day}$ , and dietary exposures were 590–1,140  $\text{ng}/\text{day}$ . The mean air exposure estimated for Springfield, Massachusetts, was almost 10 times lower (158  $\text{ng}/\text{day}$ ), while the dietary exposure (586  $\text{ng}/\text{day}$ ) was equal to the low end of the range for the population of Jacksonville, Florida. In Jacksonville, Florida, characterized as a high pesticide use area, inhalation exposure exceeded dietary exposure; in Springfield, Massachusetts, characterized as a low pesticide use area, the dietary exposure to diazinon exceeded the inhalation exposure.

Workers employed in industries that manufacture, formulate, package, or apply diazinon and workers involved in the disposal of diazinon or diazinon-containing wastes have the potential to be exposed to the highest concentrations of diazinon. In occupational settings, dermal exposure and subsequent absorption through intact skin is the most important route of exposure, and inhalation exposure is generally less important (Jeyaratnam and Maroni 1994). Inhalation of diazinon depends on its volatility, the type of

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formulation used, and the application technique employed. Occupational ingestion may occur as a result of poor work practices and/or lack of personal hygiene.

NIOSH recommends that the occupational exposure level not exceed  $0.1 \text{ mg/m}^3$  for a 10-hour TWA workday (NIOSH 2006a). In addition, the American Conference of Governmental Industrial Hygienists has recommended a time-weighted average threshold limit value (TWA-TLV) of  $0.01 \text{ mg/m}^3$  with a skin notation for occupational exposure to diazinon (RTECS 2006).

Except for professional pesticide applicators or farm workers, the exposure risks from diazinon appear relatively minor as long as label instructions are followed and safeguards are taken to avoid extensive dermal contact. Even studies of dermal exposure typical of shearers handling sheep that have been dipped in diazinon showed dermal absorption rates of  $<4\%$  (Wester et al. 1993). Contamination was generally highest on the face, hands, and arms, while inhalation was a minor route of exposure as levels of diazinon were less than a third of the TWA occupational exposure limit (Nutley et al. 1995). Studies of dermal exposure for workers in grain elevators failed to detect diazinon in grain dust above the  $0.01 \text{ } \mu\text{g/g}$  detection limit, although much higher levels have been reported from Australia (Palmgren and Lee 1984).

The use of a 2-day lag period from the time of diazinon application to the use of office or domestic indoor space appears adequate to eliminate exposure risks from vapors and residues that might be incurred from either inhalation or dermal absorption. Air sampling of a room treated with 36 pest control strips measured a maximum diazinon air concentration of  $1.34 \text{ } \mu\text{g/m}^3$  15 days post-application (Jackson and Lewis 1981). Similarly, Williams et al. (1987) found that air sampling in two animal facility areas used by facility personnel and treated monthly with a 1% aqueous diazinon solution measured  $2\text{--}3 \text{ } \mu\text{g/m}^3$   $<24$  hours post-application. Currie et al. (1990) also measured diazinon air concentrations in empty and furnished offices treated with a 1% aqueous solution. Four hours post-application, diazinon air concentrations were  $163$  and  $158 \text{ } \mu\text{g/m}^3$  in two empty offices and  $28 \text{ } \mu\text{g/m}^3$  in the furnished office. One day post-application, diazinon levels in the offices ranged from  $125 \text{ } \mu\text{g/m}^3$  (empty office) to  $27 \text{ } \mu\text{g/m}^3$  (furnished office), but by 2 days post-application, the highest diazinon air concentration measured was  $53 \text{ } \mu\text{g/m}^3$ . Air sampling levels of diazinon 2 days post-treatment in these three indoor exposure contexts were well below the NIOSH 8-hour TWA permissible exposure level (PEL) of  $100 \text{ } \mu\text{g/m}^3$ .

Residual air concentrations of diazinon in a commercial greenhouse were studied by Lenhart and Kawamoto (1994). These authors monitored diazinon air concentrations applied as a spray and by cold



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fogging. The 40-minute spray application was made to a portion of the greenhouse with only passive ventilation (adjustable window vents). During application, circulating fans were turned off and all roof vents were closed. After the spray application, 1.4 L of the diazinon emulsifiable concentrate formulation in 18 L of water was added to each of two cold fogging machines set for a 4-hour cold fogging application. Air samples were collected during the work shift prior to pesticide application, hourly during the application, and for 4 consecutive days after the pesticide application. Full shift area air samples were collected. During the post-application period, air circulating fans were continuously operated and the roof vents were open occasionally. The 8-hour TWA for the spray application ranged from not detected to  $25 \mu\text{g}/\text{m}^3$ . The 8-hour TWA diazinon concentrations were  $6.0\text{--}52 \mu\text{g}/\text{m}^3$  (Saturday),  $3\text{--}30 \mu\text{g}/\text{m}^3$  (Sunday),  $2.4\text{--}17 \mu\text{g}/\text{m}^3$  (Monday), and not detected– $12 \mu\text{g}/\text{m}^3$  (Tuesday). During the cold fogging application, diazinon concentrations on Friday were  $730\text{--}3,030 \mu\text{g}/\text{m}^3$ . Residual 8-hour TWA concentrations for this application were  $70\text{--}250 \mu\text{g}/\text{m}^3$  (Saturday),  $27\text{--}67 \mu\text{g}/\text{m}^3$  (Sunday),  $20\text{--}59 \mu\text{g}/\text{m}^3$  (Monday), and  $19\text{--}40 \mu\text{g}/\text{m}^3$  (Tuesday). Two of the 4 samples collected on Saturday exceeded the NIOSH TWA permissible exposure level of  $100 \mu\text{g}/\text{m}^3$  for occupational exposures to diazinon. Results of this study indicate that greenhouse workers can be at risk of inhalation exposure to residual diazinon concentrations. The authors believe that all diazinon applications should be conducted on Friday evenings after the greenhouse workers have left so that much of the residual pesticide can settle over the weekend.

Wright et al. (1996) conducted a study to determine the levels of insecticide residues in the ambient air of insecticide storage and office rooms in commercial pest control buildings. Diazinon was detected in the air of office rooms at levels of  $<0.01$  to  $0.36$  and  $<0.01\text{--}0.13 \mu\text{g}/\text{m}^3$  measured in summer and winter, respectively. In the air of the storage rooms, diazinon was detected at levels of  $<0.01\text{--}0.48$  and  $<0.01\text{--}0.36 \mu\text{g}/\text{m}^3$ , respectively. The mean diazinon quantity detected in the ambient air of all company rooms in 38 air samples was  $0.08 \mu\text{g}/\text{m}^3$ .

Finally, air sampling at a retail garden store conducted to determine exposures for retail employees showed levels of diazinon averaging only  $3.4 \mu\text{g}/\text{m}^3$ , well below the NIOSH TWA exposure level of  $100 \mu\text{g}/\text{m}^3$  (Wachs et al. 1983). However, these authors point out that the air concentrations they reported may vary greatly among retail stores depending on the amounts and types of diazinon formulation sold, air temperature, condition of the packaging material (e.g., torn packaging, loose lids), prior spills, and types of floor coverings.

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The National Occupational Exposure Survey (NOES) conducted by NIOSH from 1981 to 1983 estimated that 39,342 workers (including 3,216 women) employed at 3,168 facilities were potentially exposed to diazinon in the United States (NIOSH 2006b). The NOES database does not contain information on the frequency, concentration, or duration of exposure; the survey provides only estimates of workers potentially exposed to chemicals in the workplace.

**6.6 EXPOSURES OF CHILDREN**

This section focuses on exposures from conception to maturity at 18 years in humans. Differences from adults in susceptibility to hazardous substances are discussed in Section 3.7, Children's Susceptibility.

Children are not small adults. A child's exposure may differ from an adult's exposure in many ways. Children drink more fluids, eat more food, breathe more air per kilogram of body weight, and have a larger skin surface in proportion to their body volume. A child's diet often differs from that of adults. The developing human's source of nutrition changes with age: from placental nourishment to breast milk or formula to the diet of older children who eat more of certain types of foods than adults. A child's behavior and lifestyle also influence exposure. Children crawl on the floor, put things in their mouths, sometimes eat inappropriate things (such as dirt or paint chips), and spend more time outdoors. Children also are closer to the ground, and they do not use the judgment of adults to avoid hazards (NRC 1993).

In areas where diazinon is sprayed for agricultural use, children within the general population are likely to be exposed to diazinon in the same ways as adults, including through contact with sprayed plants, soil, or other surfaces; breathing contaminated air; eating contaminated foods; or drinking contaminated water. Diazinon residues bound to soil or dust particles in carpets or on bare floors may present an exposure route for infants and toddlers through dermal contact or ingestion. Translocation from household pets exposed to diazinon may also present an exposure route through dermal contact or ingestion.

Several studies suggest that house dust is an important route of toxicant exposures to young children. In a study of pesticide exposure to children in the home in rural areas in California, samples of house dust were analyzed from a day care center and 10 homes, 5 of which were also the home of at least one farm worker (currently working in the field) and 8 of which reported home pesticide use (Bradman et al. 1997). Excluding nondetects, concentrations of diazinon ranged from 0.7 to 169 mg/kg in four farm-worker homes and from 0.2 to 2.5 mg/kg in three nonfarm worker homes. Diazinon was detected at 0.10 mg/kg in the daycare center. For children in two of the homes with the highest levels of diazinon, ingestion

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exposures leading to risks for cholinesterase inhibition exceeded the EPA Office of Pesticide Program's chronic oral RfD of  $9 \times 10^{-5}$  mg/kg/day. The home with the highest level (169 mg/kg) also exceeded the EPA subchronic RfD of  $9 \times 10^{-4}$  mg/kg/day. Diazinon residues of 220, 125, and 52 ng were detected on the hands of 3 of the 11 toddlers; all 3 toddlers resided in a different farm worker home (Bradman et al. 1997). For the child with the highest diazinon level on the hands, exposures leading to risks of cholinesterase inhibition due to diazinon ingestion from hand residues also exceeded the chronic RfD (Bradman et al. 1997).

The Minnesota Children's Pesticide Exposure Study (MNCPEs) monitored house dust levels from carpets and other surfaces and hand rinses for 102 children, ages 3–13, over a 1-week sampling period (Lioy et al. 2000). Diazinon was detected in approximately 10% of carpet and surface samples collected. A different collection method yielded eight samples with detectable quantities (detection limit of 0.076 ng/cm<sup>2</sup> surface) out of 194 samples collected. It was only detected in 6 of 94 hand rinse samples collected. Seventy-five percent of the children had played on the floor and 67% had played outside between hand rinse samples, increasing their exposure to diazinon present on carpets and other surfaces.

Inhalation exposure to semivolatile pesticides in indoor air can be substantial and may be a primary route of exposure after residential use among homes using insecticides. However, an aggregate-exposure study of pesticides, including diazinon, among 102 children in Minnesota concluded that ingestion was by far the dominant route of exposure when compared to inhalation (Clayton et al. 2003). EPA regulatory action to phase out residential and some agricultural use of diazinon has been successful at reducing indoor air exposure in residential homes (EPA 2004b; Whyatt et al. 2004).

Children may be exposed to diazinon and its residues in the foods that they eat. In the FDA Total Diet Study for infants and toddlers conducted between 1980 and 1982, the average concentration and the calculated average daily intake of diazinon in different food groups were determined (Gartrell et al. 1986). In the infant diet, the average concentrations (mg/kg) and average daily intakes ( $\mu$ g/day) of diazinon by food group were 0.0002 and 0.0148 in meat, fish, and poultry, 0.0051 and 0.245 in grain and cereal products, and 0.0001 and 0.0016 in oils and fats, respectively. In the toddler diet the average concentrations (mg/kg) and average daily intakes ( $\mu$ g/day) of diazinon by food group were 0.0002 and 0.0230 in meat, fish, and poultry, 0.0034 and 0.387 in grain and cereal products, 0.0004 and 0.0268 in vegetables, 0.0001 and 0.0157 in fruit and fruit juices, 0.0007 and 0.0114 in oils and fats, and 0.0003 and 0.0073 in sugar and adjuncts, respectively. Data on the weight-adjusted intake of diazinon by infants and toddlers were determined based on the results of the FDA Total Diet Studies for fiscal years 1978–

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1981/1982 (Gartrell et al. 1986). The reported weight-adjusted intakes of diazinon ranged from 0.002 to 0.032  $\mu\text{g}/\text{kg}$  body weight/day for infants and 0.004–0.034  $\mu\text{g}/\text{kg}$  body weight/day for toddlers for the study years.

Quantitative estimates of the exposure of infants and children to pesticides have been reported in the results of FDA Total Diet Studies conducted in the 1980s using the amounts of pesticide residues in foods thought to be in the diets of infants or children. Estimates of the mean intake of diazinon per unit body weight were made for the 6–11-month-old age group, 2-year-old age group, and the 14–16-year-old female and 14–16-year-old male age groups. For the period June 1984–April 1986, the estimates were 0.0020  $\mu\text{g}/\text{kg}/\text{day}$  for the 6–11-month-old group, 0.0047  $\mu\text{g}/\text{kg}/\text{day}$  for the 2-year-old group, 0.0018  $\mu\text{g}/\text{kg}/\text{day}$  for the 14–16-year-old female group, and 0.0025  $\mu\text{g}/\text{kg}/\text{day}$  for the 14–16-year-old male group (Gunderson 1995a). For the period July 1986–April 1991, the mean daily intake estimates were 0.0061  $\mu\text{g}/\text{kg}/\text{day}$  for the 6–11-month-old group, 0.0106  $\mu\text{g}/\text{kg}/\text{day}$  for the 2-year-old group, 0.0037  $\mu\text{g}/\text{kg}/\text{day}$  for the 14–16-year-old female group, and 0.0052  $\mu\text{g}/\text{kg}/\text{day}$  for the 14–16-year-old male group (Gunderson 1995b). Diazinon residues were not detected in any of the samples of infant formula (milk-based without iron, canned, ready-to-serve) analyzed in the study (Gunderson 1995b).

In the FDA regulatory monitoring of domestic foods that may be eaten by infants/children conducted from 1985 to 1991, the maximum residue concentrations of diazinon detected were 0.46 mg/kg in apples, 0.17 mg/L in whole milk, 0.26 mg/kg in oranges, 0.06 mg/kg in pears, and trace amounts in bananas (Yess et al. 1993). In imported foods, the maximum residue concentrations of diazinon were 0.06 mg/L in apple juice, 0.08 mg/kg in apples, 0.11 mg/kg in pears, and trace amounts in bananas. In the FDA Total Diet Study of infant foods conducted over the period 1985–1991, diazinon was detected at a maximum residue concentration of 0.0004 mg/kg in infant mixed cereal, dry, prepared with whole milk, 0.0004 mg/kg in beef, high meat, and vegetables, 0.0006 mg/kg in vegetables with bacon/ham, and 0.0009 mg/kg in vegetables with beef (Yess et al. 1993). The maximum diazinon residue concentrations detected in adult foods eaten by infants/children reported in this study were 0.002 mg/kg in apples, red, with peel, raw, 0.004 mg/kg in peanut butter, creamy, and 0.005 mg/kg in pears, raw.

Based on a calculated acute population adjusted dose (aPAD), at which no adverse health effects would be expected using the safety factor prescribed in the Food Quality Protection Act (FQPA), the population subgroup with the highest acute dietary exposure (at 63% of the aPAD) and the highest chronic dietary exposure (at 22% of the cPAD) is children aged 1–6 (EPA 2004b). However, values of <100% of the

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aPAD or cPAD are considered to be not of concern (EPA 2004b). The reported aPAD and cPAD values were 0.0025 mg/kg/day (2.5 µg/kg/day) and 0.0002 mg/kg/day (0.2 µg/kg/day), respectively.

### 6.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Other than individuals who are occupationally exposed to diazinon (during its production, formulation, packaging, distribution, use, or disposal), populations exposed to higher than background concentrations of diazinon in ambient air include those living near chemical manufacturing or processing sites, individuals living on farms or in the vicinity of agricultural areas where diazinon is extensively used, and individuals living near hazardous waste sites. Individuals living near these sites may also be exposed to potentially higher concentrations of diazinon or its metabolites in their drinking water if they obtain tap water from wells located near these sources. Children may receive higher diazinon doses from dermal exposures if they play on freshly treated lawns or soil. In addition, children may receive potentially higher oral doses from ingestion of diazinon-treated soils from their hands while playing in contaminated areas. In order to mitigate the exposure and risk to the general population, especially children, the EPA has phased out all residential uses of diazinon as of December 2004 (EPA 2004b).

### 6.8 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 diazinon is available. Where adequate information is not available, ATSDR, in conjunction with 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 diazinon.

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.

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**6.8.1 Identification of Data Needs**

**Physical and Chemical Properties.** While the principal properties of diazinon are well characterized, (Howard 1991; HSDB 2008; NIOSH 2005; O'Neil et al. 2001) there are data gaps for melting point, odor and taste thresholds, autoignition temperature, and explosive limits for the compound. However, these properties are sufficient in assessing the compound's environmental fate. There are also data gaps for some spontaneously-produced degradation products, some of which may be as toxic as, or more toxic than, diazinon.

**Production, Import/Export, Use, Release, and Disposal.** According to the Emergency Planning and Community Right-to-Know Act of 1986, 42 U.S.C. Section 11023, industries are required to submit substance release and off-site transfer information to the EPA. The TRI, which contains this information for 2005, became available in May of 2007. This database is updated yearly and should provide a list of industrial production facilities and emissions.

Diazinon is commercially produced in the United States and abroad. Production in the United States was estimated to be 2.63 million kg in 1982 (HSDB 2008) and 4.67 million kg in 1990 (Larkin and Tjeerdema 2000). Recent production volume data were not located. As of June 30, 2001, manufacturing of indoor use products containing diazinon was discontinued. Manufacture of non-agricultural outdoor use products containing diazinon was discontinued as of June 30, 2003 (EPA 2004b). Production amounts of diazinon should decrease due to the discontinuation of all residential products containing this chemical. Limited data were found on import volumes; an estimated  $6.41 \times 10^4$  kg diazinon was imported into the United States in 1982 (HSDB 2008). The United States exported an estimated 5.8 million pounds of diazinon from 1997 to 2000 (Smith 2001).

Diazinon is used in agriculture as an insecticide on a variety of crops. It was formerly used extensively in home and garden applications, such as in pest strips and on turf. Due to the emerging health and ecological risks posed by diazinon, manufacturers agreed to phase out and cancel all residential products. As a result, after December 31, 2004, no diazinon products with residential uses would be registered or sold (EPA 2004b). For 1987 through 1997, total annual domestic usage of diazinon was over 13 million pounds. However, most of this (about 70%) was allocated to outdoor residential uses (EPA 2004b). Since residential uses of diazinon have been discontinued, the total annual usage should be significantly lower. Approximately 4 million pounds of active ingredient diazinon are used annually on agricultural sites (EPA 2004b).

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For ultimate disposal, large amounts of diazinon residuals should be incinerated in a unit with effluent gas scrubbing, while physical, chemical, and biological treatments may be appropriate for disposal of smaller quantities of diazinon. Composting has also been used for the disposal of diazinon-contaminated soils and organic solids (Felost et al. 2003).

**Environmental Fate.** Diazinon is moderately mobile in some soil types (Arienzo et al. 1994; Kenaga 1980; Sharom et al. 1980a). Information on the mobility of diazinon and on a major degradation product 2-isopropyl-6-methyl-4-hydroxypyrimidine in various soil types is available (Arienzo et al. 1994; Levanon et al. 1994; Sharom et al. 1980a; Somasundaram et al. 1991). In the atmosphere, diazinon is subject to degradation due to photolysis (Gore et al. 1971) and reactions with hydroxyl radicals (Glotfelty et al. 1990a; Meylan and Howard 1993; Schomburg et al. 1991; Seiber et al. 1993). In water, diazinon is subject to hydrolysis, photolysis and biodegradation. The rate of degradation of diazinon in water and soil is strongly influenced by pH (Chapman and Cole 1982; Ferrando et al. 1992; Frank et al. 1991b; Garcia-Repetto et al. 1994; Sharom et al. 1980b). Diazinon undergoes only slight photolysis in water, with reported half-life estimates ranging from 42 to 88 days (EPA 1976; Frank et al. 1991b). Diazinon can be degraded at the soil surface by photolysis (Burkhard and Guth 1979), and in soils and sediment by hydrolysis (Chapman and Cole 1982; Levanon et al. 1994; Schoen and Winterlin 1987; Sethunathan and MacRae 1969; Somasundaram et al. 1989, 1991) and by biodegradation by microorganisms (Adhya et al. 1981; Barik and Munnecke 1982; Gunner and Zuckerman 1968). Additional information on the mechanism by which diazinon is converted to diazoxon in the atmosphere would be useful; additional information on the persistence and mobility of the major degradation products of diazinon would also be useful in evaluating the environmental fate of diazinon and its degradation products.

**Bioavailability from Environmental Media.** Diazinon can be absorbed following inhalation, dermal, or oral exposures. Absorption through the skin is of major concern for exposures of farmers, farm workers, commercial applicators, or homeowners related to the use of diazinon as an insecticide or nematocide (Davis et al. 1983). Absorption via inhalation is a major concern particularly with respect to indoor exposures to diazinon within 2 days post-application of the compound as a pest control agent in commercial buildings and homes (Currie et al. 1990; Jackson and Lewis 1981; Lenhart and Kawamoto 1994; Williams et al. 1987). Additional information on the concentrations of diazinon in indoor air and in groundwater from domestic wells, particularly from environments near hazardous waste sites, is needed to determine the bioavailability of diazinon in these media.

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**Food Chain Bioaccumulation.** Diazinon has an estimated low bioconcentration potential (BCF=77) (Kenaga 1980) in aquatic organisms, which is generally confirmed by measured BCF values obtained from laboratory studies with fish and other aquatic invertebrates (El Arab et al. 1990; Keizer et al. 1991; Sancho et al. 1993; Tsuda et al. 1989, 1995). Further information on measured BCF values for additional edible fish and shellfish would be helpful, as would information on tissue residues of diazinon and its major degradation products in edible species. No information was found on studies associated with plant uptake, but diazinon is rarely detected above EPA tolerance limits (Hundley et al. 1988). Bioaccumulation in aquatic food chains does not appear to be important, and no further information on biomagnification is required.

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

Diazinon is distributed in all environmental media and has been detected in ambient air (Carey and Kutz 1985; Glotfelty et al. 1990a; Kutz et al. 1976; Lewis and Lee 1976; Schomburg et al. 1991; Seiber et al. 1993; Zabik and Seiber 1993), in indoor air (Currie et al. 1990; Jackson and Lewis 1981; Lenhart and Kawamoto 1994; Palmgren and Lee 1984; Wachs et al. 1983; Williams et al. 1987), surface water (Braun and Frank 1980; Carey and Kutz 1985; Domagalski and Kuivila 1993; Frank and Logan 1988; Frank et al. 1990a; Kendall et al. 1993; Maguire and Tkacz 1993; Pereira and Hostettler 1993; Szeto et al. 1990; Wan et al. 1994), groundwater (Cohen 1986; EPA 1989), sediment (Carey and Kutz 1985; Domagalski and Kuivila 1993; Szeto et al. 1990), and some fish (Braun and Frank 1980). The levels of diazinon in air, surface water, groundwater, and soil have been well documented. Additional information on tissue residues of diazinon and its major degradation products in edible fish and shellfish species would be particularly helpful in quantifying health risk from consumption of contaminated species.

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



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**Exposure Levels in Humans.** Data regarding levels of diazinon in humans from environmental exposures (the general population, populations living near hazardous waste sites, or occupationally exposed groups) are not available. It is arguable that these levels are not knowable because of the rapid metabolism and clearance of diazinon after it enters the body (Iverson et al. 1975; Machin et al. 1975; Mount 1984; Mücke et al. 1970). Additional studies which associate levels of diazinon in the environment and levels of diazinon metabolites in body tissues would be helpful. This information is necessary for assessing the need to conduct health studies on these populations.

**Exposures of Children.** The exposure of children to diazinon through food ingestion has been estimated fairly extensively for various age groups, including infants, toddlers, and teenagers (Gundersen 1995a, 1995b; Yess et al. 1993). Weight-adjusted intakes for these age groups have also been calculated (Gartrell et al. 1986). Studies have also been conducted to assess the extent of exposure of children to diazinon through dermal contact, inhalation, and ingestion of environmental residues (Bradman et al. 1997; Clayton et al. 2003). Data on the body burden measurements of diazinon are needed to determine exposures to children living in both agricultural and non-agricultural areas. In order to assess exposures to nursing infants, studies of breast milk contamination would be useful.

Child health data needs relating to susceptibility are discussed in Section 3.12.2, Identification of Data Needs: Children's Susceptibility.

**Exposure Registries.** No exposure registries for diazinon were located. This substance is not currently one of the compounds for which a sub-registry has been established in the National Exposure Registry. The substance will be considered in the future when chemical selection is made for sub-registries 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.

### 6.8.2 Ongoing Studies

The Federal Research in Progress (FEDRIP 2006) database provides additional information obtainable from a few ongoing studies that may fill in some of the data needs identified in Section 6.8.1.

Researchers at the Agricultural Research Service are conducting a study funded by the U.S. Department of Agriculture (USDA) to develop watershed management systems that increase productivity and improve

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water quality and ecology in the Mississippi Delta. Best Management Practice studies will be coupled with research regarding the response of aquatic organisms to potential agricultural contaminants in order to improve development of Total Maximum Daily Loads.

Researchers at the University of California are conducting a study funded by the USDA to develop a pesticide transport model to characterize the vulnerability of surface and subsurface environments to pesticide contamination. The study will quantify the migration of pesticides through soils, aquifers, and surface streams by taking into account physical, chemical, and biological processes and evaluate the potential threat of pesticides and their breakdown products to surface and subsurface systems. The model will then be applied to basins in the Sacramento-San Joaquin River and comprehensive management schemes will be identified.

Scientists at the University of California are conducting research funded by the USDA to assess the fate of hazardous chemicals in aquatic environments and their toxicity to resident organisms. The goals of the study are to characterize the dissipation and transformation processes in aquatic systems for pesticides, including diazinon, of current importance and to characterize the sublethal actions of pesticides in aquatic organisms.

Researchers at Michigan State University are conducting a study funded by the USDA to mitigate crop and turf damage caused by the Japanese beetle and the European chafer in Michigan, and ultimately in the north central United States, through host plant resistance and introduction of natural enemies. One of the objectives of the study is to find an alternative to diazinon for fall and spring control of the European chafer.