

5. POTENTIAL FOR HUMAN EXPOSURE

5. POTENTIAL FOR HUMAN EXPOSURE**5.1 OVERVIEW**

Dichlorvos is released to the environment solely by human activity. Before the mid-1950s, releases of dichlorvos were largely associated with the environmental transformation of the agricultural pesticide trichlorfon. Dichlorvos is significantly more volatile than trichlorfon or most other organophosphate pesticides. While the relatively high volatility of dichlorvos makes it impractical for application to field crops, this property makes the chemical useful as a fogger and fumigant capable of penetrating into small interstitial areas in walls and other structural surfaces to kill insects (Hayes 1982). Some of the earliest applications were around confined animal-feeding operations to control flies and other pests. By the mid-1960s, dichlorvos preparations were developed that could control the release of the vapor in a concentration range from 0.015 mg/m³ (considered the minimum needed to kill flies or mosquitoes after an exposure of several hours) to 0.25 mg/m³. This higher concentration would kill insects after exposures of half an hour or less while providing an adequate margin of safety for shortterm human exposures (Hayes 1982).

Some of the earliest applications outside animal agriculture were for the disinsection (to kill insects, mites, and other invertebrate pests) of commercial aircraft subject to quarantine requirements when crossing international boundaries (Rasmussen et al. 1963); in museum storage areas (Deer et al. 1993); and in warehouses, greenhouses, or containerized ships (Hayes 1982). By the late 1960s polyvinylchloride plastic resin strips became popular as a way to dispense dichlorvos in a time-release fashion. These strips were widely adopted in livestock holding or feeding operations and also were widely used in homes. A variety of pest strip products became popular to control insects around kitchens or other household areas, and dichlorvos strips became widely used as an ingredient in pet flea and tick collars (Hayes 1982; IARC 1991; PIP-Dichlorvos 1993). During the 1980s, these household uses were progressively reduced as the EPA began a Special Review process for existing registered uses of dichlorvos. While the Special Review was still ongoing as of the end of 1994, EPA precautionary actions by 1988 required household products containing dichlorvos to display the words "Danger-Poison." Manufacturers began switching to other pesticide agents, mostly on a voluntary basis, so that by the early 1990s, consumer use of dichlorvos had declined dramatically (EPA 1988a; Mueller 1992; PIP-Dichlorvos 1993). Its uses in agriculture and commercial buildings continue. Dichlorvos has also found applications in aquaculture to rid fish of various skin parasites (Hoey and Horsberg 1991). In

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addition, both dichlorvos and trichlorfon have veterinary and human medicinal uses in the control of severe internal and external parasite infestations (IARC 1991; PIP-Dichlorvos 1993).

Whether applied in a liquid form or dispensed through plastic resin strips or granules, dichlorvos is most commonly released to the atmosphere in a gaseous form and will predominate in the vapor phase (Eisenreich et al. 1981). Dichlorvos is not expected to undergo degradation from direct photolysis in the atmosphere (Gore et al. 1971). While degradation by ozone in the atmosphere is possible, the main degradation pathway involves the vapor phase reaction of dichlorvos with photochemically produced hydroxyl radicals. The half-life (first-order kinetics) for dichlorvos in the atmosphere has been estimated to be less than 2 days (Howard 1991; Kelly et al. 1994). Dichlorvos can be removed from the air through rainfall scavenging. When released to water, hydrolysis is the major degradation process which proceeds much more rapidly under alkaline pH conditions and with increasing temperature. The hydrolysis half-life of dichlorvos in water is highly variable and is dependent both on pH and temperature, but is typically on the order of days or weeks (Faust and Suffet 1966; Lamoreaux and Newland 1978; Lartiges and Garrigues 1995; Latif et al. 1984). Because of its high solubility in water, dichlorvos would not be expected to bioconcentrate in fish or other aquatic life. When released to soils, hydrolysis and other non-biological processes account for 70% or more of the total degradation of dichlorvos, while bacterial degradation accounts for only 30% (Lamoreaux and Newland 1978). In field studies, Menzie (1972) reported a half-life value (first-order kinetics) of 17 days for dichlorvos in soil (soil type unspecified). Because, dichlorvos does not readily sorb to soil particles, spills, or other large amounts of the pesticide, especially when released in a liquid form or dissolved in solvent carriers, may migrate through soil profiles or sediments and into groundwater. Since dichlorvos will degrade within a few days in the air, surface waters, or wetted soils due primarily to hydrolysis reactions, it is not expected to enter into large-scale regional or hemispheric fate and transport processes.

Dichlorvos has not been extensively monitored in most environmental media. It has been detected at very low concentrations in outdoor air, but has been detected at higher concentrations-in indoor air associated with its use as a fumigant pesticide in homes or during occupational exposures in manufacturing and production facilities. It has not been detected in drinking water and no environmental monitoring data were found for dichlorvos in surface or groundwater or in soils and sediment. Dichlorvos has been detected infrequently and never at concentrations of concern in both raw and processed food items.

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Since dichlorvos is a breakdown product of trichlorfon and can be generated in many plants through the metabolism of the pesticide naled (EPA 1988a; Menzie 1972; PIP-Naled 1994; PIP-Trichlorfon 1993), it is often difficult to ascertain the original source of the chemical when dichlorvos is detected in environmental media. In addition, trichlorfon and naled can be transformed to dichlorvos during sample preparation and analysis procedures. The general population may be exposed to dichlorvos primarily by inhaling contaminated air or by absorption via dermal exposure during application of liquid formulations. Both inhalation of indoor air and direct dermal contact have been identified as primary sources of human exposure at the present time. The risk of exposure is greatest to the general populations during and/or immediately after its application. Exposures of the general population to dichlorvos via consumption of contaminated food and drinking water are negligible to insignificant. Because it is rapidly metabolized, dichlorvos has not been detected in blood, adipose tissue, breast milk, or any other tissue samples from the general population or from populations with occupational exposures. In occupational settings, exposure is via inhalation of contaminated air and/or via dermal contact with dichlorvos formulations.

Dichlorvos has been identified in at least 3 of the 1,428 current or former hazardous wastes sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HAZDAT 1996). However, the number of sites evaluated for dichlorvos is not known. The frequency of these sites within the United States can be seen in Figure 5-1.

5.2 RELEASES TO THE ENVIRONMENT

Releases of dichlorvos are required to be reported under the Superfund Amendments and Reauthorization Act (SARA) Section 313; consequently, data are available for this compound in the Toxic Release Inventory (TRI) (EPA 1995a). According to the TRI, in 1993 a total of 1,562 pounds (709 kg) of dichlorvos were released to the environment from 5 large processing facilities (TR193 1995). In addition, an estimated 4,660 pounds (2,114 kg) were transferred off-site (TR193 1995). Table 5-1 lists these releases. The TRI data should be used with caution because only-certain types of facilities are required to report (EPA 1995a). This is not an exhaustive list.

Dichlorvos has been identified in a variety of environmental media collected at 3 of the 1,428 former or current NPL hazardous waste sites (HazDat 1996).

Figure 5-1. Frequency of NPL Sites with Dichlorvos Contamination

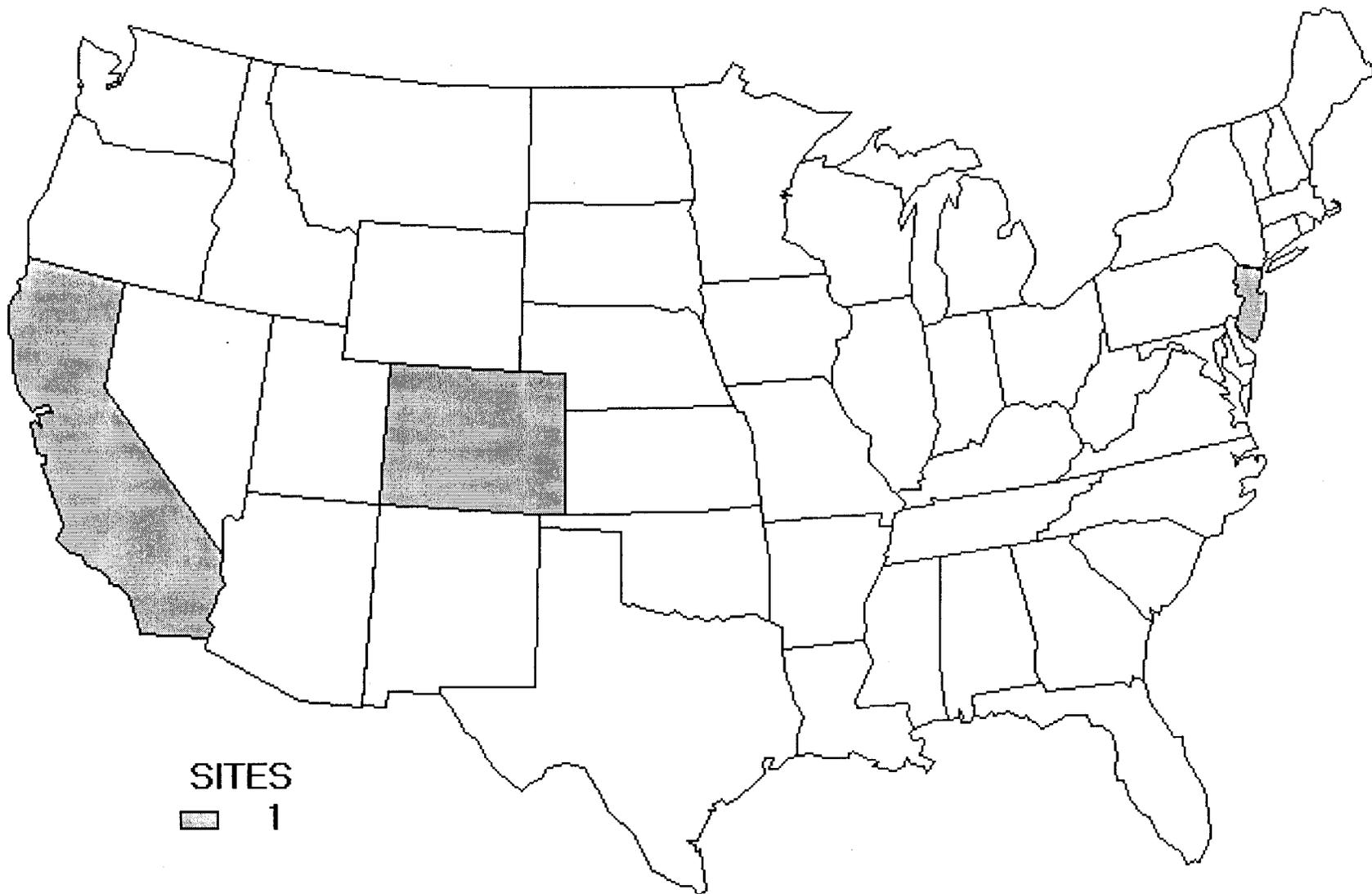


Table 5-1. Releases to the Environment from Facilities That Manufacture or Process Dichlorvos

State ^a	City	Facility	Reported amounts released in pounds per year						
			Air	Water	Land	Underground injection	Total environment ^b	POTW transfer	Off-site waste transfer
CA	LOS ANGELES	AMERICAN VANGUARD CO.	250				250		1,005
GA	SANDERSVILLE	NA	52				52		
IA	PLEASANTVILLE	NA	250				250		
KS	ELWOOD	NA	255	5			260		3,655
TX	ADDISON	NA	500		250		750		
Totals			1,307	5	250		1,562		4,660

Source: TRI93 1995

^a Post office state abbreviations used

^b The sum of all releases of the chemical to air, land, water, and underground injection wells by a given facility

NA = not available; POTW = publicly owned treatment works

5.2.1 Air

Dichlorvos may be released to the air during production and processing activities or during its various use applications as a pesticide. Based on its vapor pressure of 1.2×10^{-2} mm Hg at 20°C (see Table 3-2), a large portion of the dichlorvos contained in plastic or resin strips, or applied in concentrated liquid formulations to surfaces within buildings or other enclosures with low humidities would be expected to volatilize to the air. However, no data were located providing comprehensive estimates of releases to the air.

According to the TRI, in 1993, releases of 1,307 pounds (593 kg) dichlorvos to the air from 5 large processing facilities accounted for about 84% of the estimated total environmental releases (TR193 1995). Table 5-1 lists amounts released from these facilities. The TRI data should be used with caution because only certain types of facilities are required to report (EPA 1995a). This is not an exhaustive list.

Dichlorvos has not been found in air samples collected at any of the 3 NPL sites where it was detected in some environmental media (HazDat 1996).

5.2.2 Water

Dichlorvos may be released to surface water in waste waters generated by its production or fabrication into resin strips or other formulations. Some releases to surface water would also occur when dichlorvos is used in aquaculture to control fish parasites. Other releases to water might take place as the result of spills or other accidents. No data were found providing comprehensive estimates of intentional and unintentional releases into water.

According to the TRI, estimated releases of 5 pounds (2.3 kg) of dichlorvos to surface water from one large processing facility in 1993, accounted for about 0.3% of the estimated total environmental releases (TR193 1995). Table 5-1 lists these releases. The TRI data should be used with caution because only certain types of facilities are required to report (EPA 1995a). This is not an exhaustive list.

Dichlorvos has been found in groundwater samples (concentrations unspecified) at 1 of the 3 NPL sites where it was detected in some environmental medium (HAZDAT 1996).

5.2.3 Soil

Dichlorvos may be released to the soil as a result of intentional uses, spills or through waste disposal practices associated with its various formulations. According to the TRI, estimated releases of 250 pounds (113 kg) of dichlorvos to the land from one large processing facilities in 1993 accounted for about 16% of the total environmental emissions. In addition, over 4,660 pounds (2,113 kg) were disposed of via offsite waste transfer (TR193 1995). The TRI data should be used with caution because only certain types of facilities are required to report (EPA 1995a). This is not an exhaustive list.

Environmental releases of dichlorvos to agricultural soils have occurred from its intentional use in livestock feeds to control internal parasites and flies (IARC 1979, 1991; OSU 1992) and its use in specialty crops (e.g., tobacco and greenhouse-grown food crops) (IARC 1991). Land application of the treated livestock manures may contribute to soil releases of dichlorvos (OSU 1992). Any improper disposal of plastic resin material wastes containing dichlorvos or other formulations of dichlorvos could also release the chemical to soils. The most recent use application data from the mid- to late 1980's suggests that agricultural applications of dichlorvos represented 60% of the total annual use of dichlorvos in the United States. Commercial, institutional, and industrial uses accounted for 25% of annual usage; while domestic uses, including household pest control and pet collars accounted for only 15% of annual usage (IARC 1991). Releases of dichlorvos to soils in urban areas have occurred from its intentional uses in residential, industrial, and commercial areas including food services, food manufacturing and processing facilities; in horticultural and turf applications for pest control; and in flea collars, room foggers, and no-pest strips. For products composed of polyvinylchloride plastic resins impregnated with dichlorvos, such as flea collars, the collar materials were designed to shed off small fragments of the plastic material (exfoliation). These small particles tend to spread throughout the animal's coat, which would assist in delivering the pesticide to fleas and other insect parasites. Such small particles could also be spread onto floors or land surfaces, where they could be released to the soil. Drastic changes in recent use patterns (EPA 1993a) and restrictions on or cancellations of some registrations (EPA 1991a) make it difficult to determine the magnitude of current uses/applications of dichlorvos are greater in both agricultural and in urban settings

Dichlorvos has been found in soil samples (concentrations unspecified) at 1 of the 3 NPL sites where dichlorvos was detected in some environmental media (HAZDAT 1996).

5.3 ENVIRONMENTAL FATE

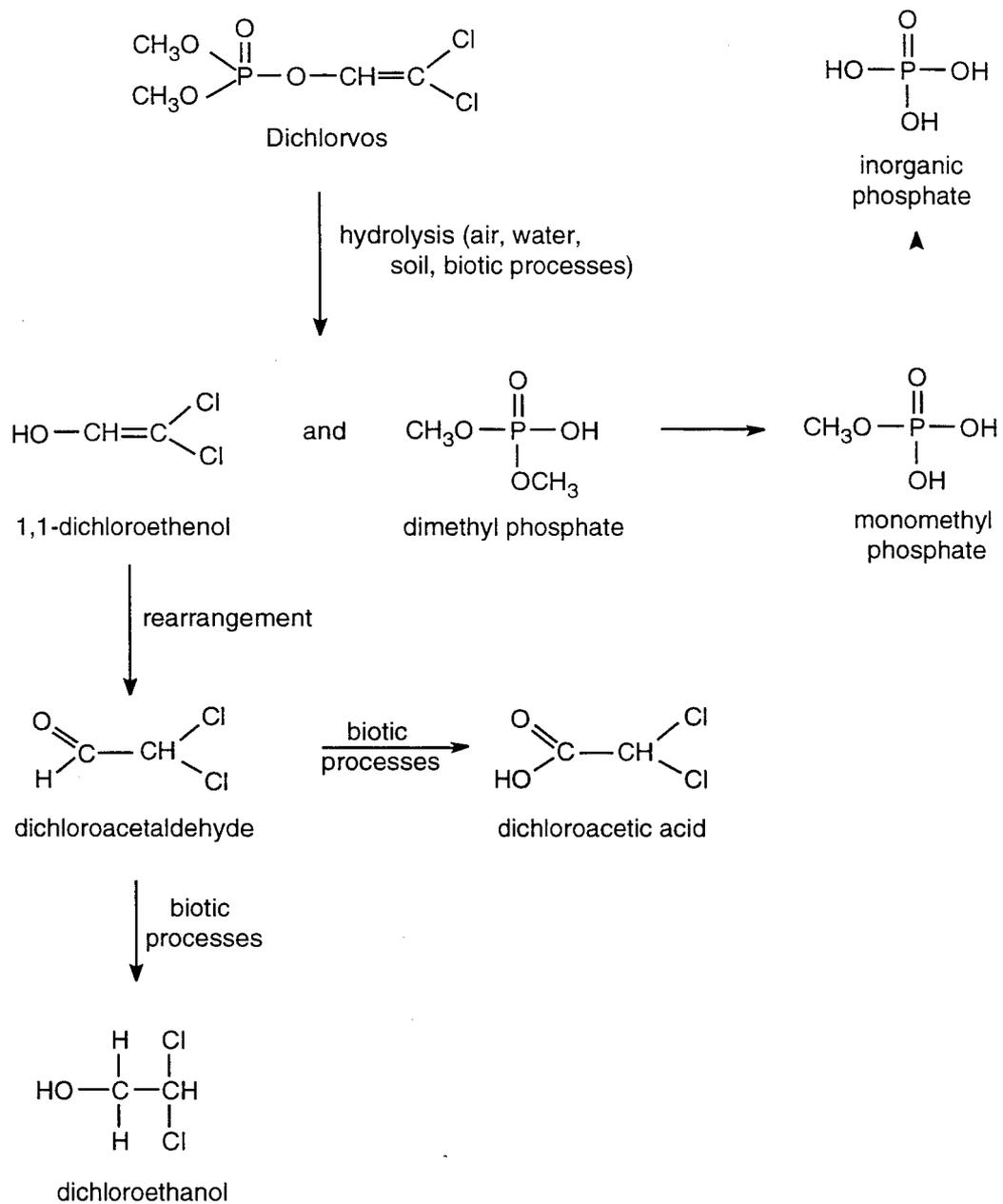
While most initial releases of dichlorvos will be to the air, the solubility of dichlorvos in water will quickly lead to its partitioning to water. Once in contact with water, hydrolysis reactions become the predominant mechanisms for degradation, with hydrolysis proceeding more rapidly with increasing pH and temperatures (Faust and Suffet 1966; Lamoreaux and Newland 1978). In the environment, abiotic degradation via hydrolysis is the primary transformation process, although biodegradation by some microorganisms also occurs (see Figure 5-2). Dichlorvos shows little tendency to sorb to soil particles or bioconcentrate in living tissues, which creates the potential for dichlorvos to leach through soil and sediments into groundwater (PIP-Dichlorvos 1993).

5.3.1 Transport and Partitioning

Dichlorvos has a high vapor pressure (1.2×10^{-2} mm Hg) and if released to the atmosphere, it will be found predominately in the vapor phase (Eisenreich et al. 1981). Most dichlorvos releases are initially intended to reach the air, but if there is high humidity or rain scavenging, dichlorvos will partition into water. The Henry's law constant for dichlorvos has been calculated as 7.01×10^{-8} atm-m³/mole at 25°C (see Table 3-2), which is a fairly low value and suggests that dichlorvos, once dissolved in water, will remain in aqueous solution for an appreciable period of time. The solubility of dichlorvos in water is 16,000 mg/L (Table 3-2), which is quite high compared to other common organophosphate pesticides (e.g., <10 mg/L for malathion or parathion and <100 mg/L for diazinon). Based on the Henry's law constant and for aqueous systems with initial dichlorvos concentrations less than the saturation levels, the volatilization half-life from a model river was estimated at 57 days, while the volatilization half-life from a model pond was estimated at over 400 days (Howard 1991).

The log K_{oc} (organic carbon partition coefficient) for dichlorvos has been calculated to be 1.45 (Kenaga 1980), which is very low. This K_{oc} value suggests dichlorvos will not appreciably sorb to soil particles; therefore, dichlorvos can be expected to leach through soil columns or sediments and into groundwater (Swann et al. 1983). Since natural soils and sediments contain water, hydrolysis reactions will also occur as the dichlorvos moves downward through the soil horizon, and this will reduce the amount of dichlorvos transported to groundwater.

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Figure 5-2. Environmental Transformation for Dichlorvos in Soil and Sediment

Because of its high water solubility, dichlorvos would not be expected to show a significant tendency to bioconcentrate in fish or other aquatic animals. A commonly used quantitative measure of the tendency of some chemicals to partition from the water column into the tissues of aquatic organisms is the bioconcentration factor (BCF). This BCF value can be empirically determined by measuring the chemical concentration in an organism's tissues after exposing it over a period of time to a specified concentration of the chemical in water. The empirical BCF is calculated as the concentration of the chemical in an organism divided by the concentration of the chemical in water. BCFs can sometimes be estimated based on regression equations developed from data on physical properties such as the octanol/water partition coefficient and the water solubility. The larger the value of the BCF, the greater the tendency for partitioning into the tissues of an organism. No experimental BCF values for dichlorvos have been reported for any aquatic species. A calculated BCF of 2.8 has been reported (Kenaga 1980), which is intended to be applicable to many fishes. More recently, the EPA's ASTER Toxicology profile database reported a calculated BCF of 1 for the fathead minnow (*Pimephales promelas*) (ASTER 1996). Such BCF values suggest there is no marked tendency for bioaccumulation or biomagnification of dichlorvos through the food chain. One short-term exposure study was conducted to test the efficacy of dichlorvos as a pest control agent for sea lice on penned salmon. Atlantic salmon (*Salmo salar*) were exposed to dichlorvos at 2 ppm at either 4 or 12 °C for 60 minutes as a treatment to remove ectoparasitic copepods (Horsberg and Hoy 1990). These authors found that at 4 °C, dichlorvos residues averaged 0.094 µg/g (ppm) in muscle and <0.01 µg/g in liver tissue immediately post-treatment. At 12 °C, dichlorvos residues averaged 0.046 µg/g (ppm) in muscle and 0.041 µg/g (ppm) in liver tissue. At 4 °C, residues in muscle and liver were not detected after 3 days post-treatment and at 12 °C, residues were not detected in muscle tissue after 1 day post-treatment and in liver tissue 6 days post-treatment. The rapid clearance of dichlorvos from the fish tissues was similar to results obtained in other species (see Chapter 2).

5.3.2 Transformation and Degradation

5.3.2.1 Air

Dichlorvos does not strongly absorb ultraviolet (UV) light above 240 nm (Gore et al. 1971; Howard 1991) and is, therefore, unlikely to be subject to direct photolysis in the atmosphere. Based on theoretical considerations, dichlorvos may be susceptible to degradation in the air from free radicals such as hydroxyl groups or ozone. Figure 5-2 shows the transformation pathways for dichlorvos in the

atmosphere. A calculated half-life, assuming first-order kinetics, for a vapor-phase reaction of dichlorvos with photochemically produced hydroxyl radicals was reported to be 2 days, assuming an atmospheric concentration of 5×10^5 hydroxyl radicals per cm^3 (Howard 1991). With an assumed level of 7.0×10^{11} ozone molecules per cm^3 , an estimated half-life (first-order kinetics) for dichlorvos of 320 days was reported (Howard 1991).

5.3.2.2 Water

Dichlorvos is highly soluble in water and tends to remain in solution, with very little tendency to sorb to sediments. While dichlorvos can re-volatilize to the air, re-volatilization of dichlorvos once it is dissolved in water is a fairly slow process.

When dissolved in water, dichlorvos becomes subject to both abiotic and biological degradation. The predominant degradation mechanism is hydrolysis, and dichlorvos is hydrolyzed into dichlorethanol, dichloroacetaldehyde, dichloroacetic acid, dimethyl phosphate and dimethyl phosphoric acid (WHO 1989). The hydrolysis reaction is sensitive to pH, with much more rapid degradation taking place at higher, more alkaline pH levels. Chemical hydrolysis of dichlorvos in aqueous, buffered, soil-free systems showed that hydrolysis did not occur in very acid systems ($\text{pH} < 3.3$), but increased with increasing pH values (26% in 4 days at $\text{pH} 6.9$), and was most rapid at $\text{pH} 9.3$ (>99% in 2 days). At pH values of 2.0, 3.3, 6.2, 6.9, 7.8, 8.2, 8.7, and 9.3, the percent of initial dichlorvos remaining in buffered, aqueous solutions after 96 hours was 99, 100, 92, 89, 84, 82, 36, and 10%, respectively (Lamoreaux and Newland 1978). Latif et al. (1984) reported half-life (first-order kinetics) values for dichlorvos of 4,620 minutes (3.2 days), 2,100 minutes (1.46 days), 462 minutes (0.32 days), and 301 minutes (0.2 days) in water at $\text{pH} 5.4$, 6, 7, and 8, respectively.

In addition to pH, hydrolysis kinetics are affected by temperature. Faust and Suffet (1966) reported that the half-life (first-order kinetics) values for dichlorvos in water were 240, 61.5, 17.3, 1.7, and 0.2 days at 10, 20, 30, 50, and 70 °C, respectively. Lartiges and Garrigues (1995) studied degradation of dichlorvos in different types of water (filtered water, natural seawater, natural river water, and filtered river water) under different environmental conditions (temperatures of 6 and 22 °C to simulate winter and summer conditions, respectively, and pH values ranging from 6.1 to 8.1). The authors reported that at the winter temperature, dichlorvos was still present in the filtered water ($\text{pH} 6.1$) after 180 days. Residues in river water ($\text{pH} 7.3$) and filtered river water ($\text{pH} 7.3$) disappeared after 81 days

and residues in seawater (pH 8.1) disappeared after 34 days. At summer temperatures, dichlorvos residues disappeared in filtered water after 81 days. Residues in river water disappeared after 55 days, residues in filtered river water disappeared in 34 days, and residues in seawater persisted for 180 days. At pH values of 7.8-8.2, reported half-disappearance times were approximately 96 and 48 hours, respectively (Lamoreaux and Newland 1978). At pH values of 8.7 and 9.3, the half-disappearance time was less than 24 hours and at pH values of less than 6.9, the half-disappearance time was more than 96 hours. Therefore, under pH conditions encountered in most natural waters in contact with the atmosphere, hydrolysis can degrade dichlorvos at rates considerably more rapid than the rate of dichlorvos re-volatilization to the air. Only at low pHs (less than 4) would dichlorvos show persistence. The hydrolysis half-life or half-disappearance time in water is highly variable and is dependent both on pH and temperature, but is typically on the order of days to weeks.

Microorganisms found in sewage sludges apparently can biodegrade dichlorvos, but a period of acclimation may be needed and the rate of biodegradation may be much less than that from abiotic transformation processes (Lieberman and Alexander 1983). Abiotic hydrolysis rates will generally far outstrip the biodegradation rate, and the biotic transformations will be sensitive to pH, temperature, and the concentrations of dichlorvos or other toxic chemicals present. The environmental transformation pathways for dichlorvos in water are shown in Figure 5-2.

Since the hydrolysis reaction in water is both pH- and temperature-dependent, caution should be exercised in making predictions for dichlorvos in groundwater. In groundwater with low pH (pH <4), dichlorvos would be expected to persist for moderate periods, with half-disappearance times greater than 4 days (PIP-Dichlorvos 1993).

5.3.2.3 Sediment and Soil

In soils containing any appreciable amount of moisture, hydrolysis reactions similar to those in aqueous solutions are expected to occur (PIP-Dichlorvos 1993), although few quantitative studies of half-life or half-disappearance times could be identified in the literature. In field trials, Menzie (1972) reported a half-life (first-order kinetics) value for dichlorvos of 17 days; however, the soil type was not specified by the author. In laboratory experiments using natural and sterilized Houston black clay soils, the rate of dichlorvos disappearance in the soil was related directly to the presence of the bacterium *Bacillus cereus*, the pH of the soil perfusion system, and the extent of dichlorvos adsorption.

Dichlorvos disappearance was most rapid when *B. cereus* was added to a previously unsterilized soil (half-life value of 3.9 days), while under sterile conditions, a half-life value of 10 days was reported (Lamoreaux and Newland 1978). Hydrolysis and other non-biological processes accounted for 70% of the total degradation of dichlorvos, while bacterial degradation accounted for only 30% in the soil perfusion system (Lamoreaux and Newland 1978).

For sediments, biodegradation processes may often be similar to the results documented for sewage sludges. A sewage sludge culture containing the bacterium *Pseudomonas aeruginosa* and other bacteria from the same genus was shown to convert dichlorvos into several metabolites, including dichloroethanol, dichloroacetic acid, ethyl dichloroacetate, and inorganic phosphate (Lieberman and Alexander 1983). Dichlorvos was still present in the cultures after 7 days of incubation at 29 °C both in the presence or absence of microbes. The environmental transformation pathways for dichlorvos in soil and sediment are shown in Figure 5-2. Abiotic hydrolytic cleavage of the parent molecule presumably gives rise to dichloroacetaldehyde (via rearrangement of 1,1-dichloroethanol) and dimethyl phosphate. The dichloroacetaldehyde then is converted enzymatically to dichloroethanol, or to dichloroacetic acid (Lieberman and Alexander 1983). The dimethyl phosphate is converted to inorganic phosphate with monomethyl phosphate being a likely intermediate.

5.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

Dichlorvos has not been routinely included in most national environmental monitoring programs, so that broad-based data are lacking for most environmental media. Results from several special studies suggest that dichlorvos should not be found above very low levels on a widespread basis in either air or in water. Data for soils and sediments are even more limited. When dealing with soils and sediments, a complicating factor is that dichlorvos may be found associated with fragments of discarded resin strips or in other plastics wastes or containers impregnated with dichlorvos.

Where dichlorvos is detected in the environment, it may be impossible to determine whether the initial release was from dichlorvos directly or from the application of the pesticide trichlorfon which can degrade to form dichlorvos or the pesticide naled which can be transformed in some plant tissues to dichlorvos. In addition, dichlorvos can be produced from trichlorfon and naled during sample preparation and analysis (see Section 6.2). For these reasons, caution must be used in the interpretation of data from environmental samples found to contain dichlorvos. Reliable evaluation of

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

5.4.1 Air

Most of the available measurements of dichlorvos in the air have focused on indoor-air settings where dichlorvos has been most commonly used as a fumigant-type pesticide agent; however, limited information is available for ambient outdoor air. Kelly et al. (1994) reported that dichlorvos concentrations measured in 288 samples of ambient outdoor air collected from 2 locations ranged from not detected to 150 ng/m³. As part of the Non-Occupational Pesticide Exposure Study based on the EPA's Total Exposure Assessment Methodology (TEAM) approach, dichlorvos was monitored in outdoor air samples collected in Springfield, Massachusetts and in Jacksonville, Florida. While dichlorvos was not detected in ambient outdoor air in Springfield, Massachusetts, it was detected at 3.2 ng/m³ in winter samples collected in Jacksonville, Florida (Whitmore et al. 1994).

Much of the historic indoor-air monitoring literature deals with product uses that have declined dramatically, or ceased altogether, since the late 1980s. For instance, air monitoring data were gathered while dichlorvos was a major ingredient in pest strips which were widely used in homes, hospitals, restaurants, and other public places to kill flies, cockroaches, and other nuisance insects (Elgar and Steer 1972; Leary et al. 1974). Other valuable information comes from studies on potential occupational exposure to personnel in museums where dichlorvos was used to kill insects in closed display cases or storage areas (Deer et al. 1993), exposures to employees at professional pest control companies (Wright and Leidy 1980), and exposures to flight crews while it was used as a disinsection agent (prophylactic insecticide) in commercial aircraft (Rasmussen et al. 1963).

By the late 1960s the design of resin strips or application rates of dichlorvos in liquid or fogger formulations was guided by the goals of achieving dichlorvos concentrations in the air of at least 0.015 mg/m³ for a period of several hours; this goal to achieve good kills of nuisance insects was balanced against the rule of thumb that concentrations below 0.25 mg/m³ provided an adequate margin of safety for human health (Hayes 1982). Where adequate precautions are observed, and barring

accidents. concentrations above this 0.25 mg/m^3 threshold are uncommon in studies accompanied by good sampling programs.

Results from early work on the use of dichlorvos-impregnated polyvinylchloride resin pest strips for insect control in homes in the United Kingdom, Australia, and France suggest that indoor-air concentrations in houses that are well ventilated (especially using open windows) will be very low (range, $0.01\text{-}0.24 \text{ }\mu\text{g/L}$ [mg/m^3]) (Elgar and Steer 1972). Leary et al. (1974) monitored the concentrations of dichlorvos in indoor-room air in homes in Arizona using polyvinylchloride resin strips containing dichlorvos. Homes were treated at a rate of one strip per 1,000 cubic feet and typically averaged 8-18 strips per residence. One month after installation, new pest strips were placed in the kitchen and dining areas at a rate of one strip per 500 cubic feet. Air samples were collected prior to installation of the strips and at 1, 2, 3, 4, 6, 7, 10, 13, 16, 21, and 28 days post-application. Dichlorvos concentrations in the air peaked at $0.12\text{-}0.13 \text{ mg/m}^3$ within several days and then declined to a plateau level of $0.08\text{-}0.09 \text{ mg/m}^3$ from day 12 to 28 post-application. The replacement and doubling of the number of strips in the dining room and kitchen produced a maximum concentration of 0.16 mg/m^3 within 2 days and a subsequent decrease to 0.11 mg/m^3 by day 15. Removal of all the pest strips from the residence resulted in a rapid decline in dichlorvos residues to zero within 17 days.

In the EPA TEAM study of indoor-air levels of common household pesticides, dichlorvos was included as a target analyte, but was not commonly detected in the indoor air of the homes sampled (detected in only 1 out of 9 homes sampled) (Lewis et al. 1988). Most recently, as part of the Non-Occupational Pesticide Exposure Study based on the EPA's TEAM approach, dichlorvos was monitored in indoor-air samples collected in Springfield, Massachusetts and in Jacksonville, Florida. Dichlorvos was detected in ambient indoor air in Springfield, Massachusetts at concentrations of 4.3 and 1.5 ng/m^3 in spring and winter air samples, respectively. This pesticide was detected at much higher concentrations of 134.5 , 86.2 , and 24.5 ng/m^3 in summer, spring, and winter samples, respectively, collected in Jacksonville, Florida (Whitmore et al. 1994).

The highest occupational exposure reported was 3 mg/m^3 (mean concentration of 0.7 mg/m^3) that was monitored in a vaporizer production plant and its packaging rooms (Menz et al. 1974 in IARC 1991). Other valuable information comes from studies on potential occupational exposure to personnel in museums where dichlorvos was used to kill insects in closed display cases or storage areas (Deer et al. 1993), exposures to employees at professional pest control companies (Wright and Leidy 1980), and

exposure to flight crews while it was used as a disinsection agent in commercial aircraft (Rasmussen et al. 1963). Air samples were collected in several locations within a museum of natural history's entomological curation area (Deer et al. 1993). The authors reported a significant difference in the mean concentrations of dichlorvos in the air collected with the ventilation system on (0.005 mg/m^3) and the ventilation system off (0.14 mg/m^3). Ambient air in the storage rooms (where concentrates of various chemicals were stored and mixed to appropriate dilutions) and main office areas of four North Carolina commercial pest control firms were monitored for dichlorvos for a 4-hour sampling period (Wright and Leidy 1980). These authors found that concentrations in the storage rooms averaged 617 ng/m^3 (range, $147\text{-}1,501 \text{ ng/m}^3$) and concentrations in the office areas averaged 41 ng/m^3 (range, $19\text{-}66 \text{ ng/m}^3$). In addition, mean dichlorvos concentrations of 110 ng/m^3 (range, $16\text{-}231 \text{ ng/m}^3$) were detected in the ambient air inside company pickup trucks and vans used by professional pest control technicians (Wright and Leidy 1980). The proposed use of dichlorvos as an aircraft disinsectant was studied by Rasmussen et al. (1963). Volunteers were exposed to dichlorvos at concentrations varying from 0.14 to $0.33 \text{ } \mu\text{g/L}$ (mg/m^3) in a setting designed to simulate the maximal exposure that airline flight crew personnel could possibly receive during the maximum period of scheduled duty flights (39 doses in a 14-day period).

5.4.2 Water

Dichlorvos was included as one of 101 pesticide target analytes in EPA's National Pesticide Survey of Drinking Water Wells (EPA PIN 1994). This survey included wells from community systems used in towns and cities, non-community systems found in such public places as truck stops, and private domestic wells. Dichlorvos was not found above detection limits in any of the drinking water wells sampled. While dichlorvos is not ordinarily included in sampling programs for community systems that rely on surface water supplies, the nature of the treatment processes used in most community systems tends to minimize the possibilities for dichlorvos to persist in the final tap water. In most larger community drinking water systems in the United States that use surface water supplies, conventional treatment technologies using lime and soda ash will temporarily raise the alkalinity ($\text{pH} > 9$). These elevated pHs, combined with the ample opportunities for hydrolysis degradation before the treated water reaches end users, virtually eliminate any exposure possibilities to dichlorvos in treated drinking water.

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No other information was found on the concentrations of dichlorvos in surface water or groundwater in the United States. While dichlorvos could be released to surface waters during production or on an episodic basis as a result of spills, prompt sample collection and analysis would likely be needed to document the presence of dichlorvos. For instance, a dichlorvos formulation known as Nuvan® is widely used in Europe to control skin parasites in fish culture, especially in the caged cultures of Atlantic salmon. In a coastal bay in Ireland, monitoring was carried out during the summer period when salmon were being tended in cages in the bay and for a period after the summer to document the possible persistence of dichlorvos in seawater and sediments (Tully and Morrissey 1989). In all but one of the ambient water samples from the Beirtreach Bay study, dichlorvos levels were below the detection limit of 0.02 µg/L (ppb), suggesting that dichlorvos rapidly dissipates or degrades in large natural waterbodies. The exceptional seawater sample in which the pesticide was detected, contained 0.13 µg/L (ppb) of dichlorvos.

5.4.3 Sediment and Soil

No information was found on background levels of dichlorvos in either agricultural or urban soils or in sediments in the United States. Given the relatively high volatility of dichlorvos combined with its low potential to sorb to soil particles, and its rapid degradation via hydrolysis, dichlorvos does not appear to be a good candidate analyte for soil or sediment sampling programs.

Dichlorvos has been identified at unspecified concentrations in soil samples at 1 of the 3 NPL sites where it was detected in some environmental media (HazDat 1996).

5.4.4 Other Environmental Media

Dichlorvos has been detected in some foods as part of standard FDA residue monitoring or regional or state sampling activities using protocols patterned after the FDA methods. FDA regulatory surveillance monitoring extends back to the 1970s. The food items sampled include common fruits and vegetables, coffee, wine, and milk. In summaries of the FDA pesticide residue monitoring of foods, dichlorvos is reported among those chemicals where some detections were documented, but never at concentrations of concern. Positive samples have been detected for some food items for which tests were performed in the FDA program from 1978 to 1982 (Yess et al. 1991a), from 1983 to 1986 (Yess et al. 1991b), for 1987 (FDA 1988), 1988 (FDA 1989), 1989 (FDA 1990), 1990 (FDA 1991), 1991 (FDA 1992), 1992 (FDA 1993), 1993 (FDA 1994), and 1994 (FDA 1995). For chemicals

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with such a low incidence of detection, the published literature does not generally record concentrations detected, the food items in which the residues were detected or detection limits. Concentrations of dichlorvos in ready-to-eat foods were monitored for 10 years from 1982 through 1991 through the FDA Revised Market Basket Survey (KAN-DO Office and Pesticide Team 1995). Dichlorvos was detected in one sample of one food item (rye bread) at a concentration of 0.01 µg/g ppm.

A summary of the results of chemical residue monitoring conducted by 10 state food laboratories from 1988 to 1989 showed 2 detections of dichlorvos out of 13,085 food items sampled (0.015% positive detections), with neither of these detections exceeding any federal or state tolerance limits (Minyard and Roberts 1991). A summary of food residue sampling by the State of California included dichlorvos in its screening list, but no detections were reported for testing carried out in 1989 (Okumura et al. 1991). A special study on pesticide residues in infant formulas carried out for the National Academy of Sciences included dichlorvos as a target analyte, but no detections were reported based on analysis of 24 milk-based and 19 soy-based infant formula samples (Gelardi and Mountford 1993). Very low percentages of detections and concentrations well below any thresholds of concern are typical of food residue monitoring programs in Canada and the United Kingdom (IARC 1991).

While detections have been documented in food surveys, dichlorvos residues in food are normally destroyed by washing and cooking (IARC 1991); therefore, the risk to the general population in the United States is insignificant. In a study from Japan, residues of dichlorvos were found on harvested rice, where dichlorvos and other pesticide or fumigant agents had been applied to control insects during shipping and storage (Nakamura et al. 1993). The levels of dichlorvos decreased during storage, falling to below detection levels within 45 days after the initial treatments. In addition, ordinary washing and cooking to prepare the rice for human consumption essentially removed all of the dichlorvos residues.

5.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

While no quantitative information is available on the percentage of dichlorvos released to each environmental compartment, dichlorvos can be released to any or all environmental media (air, surface water, groundwater, and soil). The general population is exposed to dichlorvos primarily through inhalation and dermal contact. Exposure to dichlorvos via consumption of contaminated food is

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insignificant because daily intakes are extremely low, and washing and general food preparation procedures further remove dichlorvos residues (IARC 1991; Nakamura et al. 1993). Based on the results of the 2 recent FDA Total Diet Studies conducted in 1991 and 1992, the estimated mean daily dietary intakes of dichlorvos were <0.0001 , <0.0001 , and 0.0001 $\mu\text{g}/\text{kg}$ body weight for 6-11-month-old infants, 14-16-year-old males, and 60-65-year-old women, respectively (FDA 1992). Based on the results of the recent analysis of data from the FDA Total Diet Studies conducted from 1986 to 1991, the estimated mean daily dietary intakes of dichlorvos were <0.0001 , <0.0001 , and <0.0001 $\mu\text{g}/\text{kg}$ body weight for 6-11-month-old infants, 14-16-year-old males, and 60-65-year-old women, respectively (FDA 1993). Exposure to dichlorvos via consumption of contaminated drinking water also is insignificant.

The general population is exposed to dichlorvos primarily through inhalation of contaminated indoor air either during and/or immediately after application or through use of polyvinylchloride resin strips. Exposure would be of most concern in buildings with inadequate or defective ventilation. Since many commercial and residential buildings are sprayed with dichlorvos formulations or use pest control strips that vaporize dichlorvos, there is the possibility of widespread low-level exposure to many individuals in the general population from inhalation of residual vapors in these dichlorvos-treated indoor-air spaces. The second major route of exposure to dichlorvos for the general population is through direct dermal contact with the chemical spray during domestic applications, contact with dichlorvos-treated plant materials such as grass or ornamental plants, or contact with other treated surfaces (e.g., furniture) in domestic or office buildings. Through the late 1980s the widespread use of dichlorvos in resin strips for the control of insects within houses or other public buildings created the potential for widespread low-level exposures to many individuals in the general population. After 1988, concerns over the carcinogenic potential of dichlorvos led the EPA to require warning labels on pest strips. The FDA then disallowed the use of these products in kitchens, restaurants, or other places where food is prepared. This led to a dramatic reduction in dichlorvos formulations for home use. As of September 28, 1995, the EPA proposed cancellation of dichlorvos for all home uses (pest strips, flea collars, room foggers, and on ornamental lawns and plants) and for many commercial and industrial uses (TOXLIST 1995). In addition, the EPA proposed that most retained uses be restricted to specially trained certified applicators. The current potential for exposure of the general population to dichlorvos appears to be very limited compared to past exposures. In addition, if the proposed cancellation of all home use and many commercial and industrial uses of dichlorvos becomes final, this will further reduce risks to the general population.

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Dichlorvos may still be applied by professional exterminators for insect control in buildings (Berteau et al. 1989; Gold and Holcslaw 1985; Wright and Leidy 1980) and in turf grass treatments (Goh et al. 1986). These applications create potential for some exposure for the general population through inhalation, dermal contact, and (especially in children) oral intake. These exposure risks would be highest for periods immediately following dichlorvos applications and would be of most concern in buildings with inadequate or defective ventilation (Gold and Holcslaw 1985).

Because dichlorvos is metabolized so rapidly by esterases in the liver and blood, it has not been found in human adipose tissue, milk, blood or in urine (see Section 2.5). Even the major metabolic products of dichlorvos, dimethyl phosphate and the glucuronide conjugate of dichloroethanol, are rapidly excreted from the body and will have left the body completely within a day or two of cessation of dichlorvos exposure.

Although many of the uses of dichlorvos have been restricted, occupational exposures may still occur among workers at facilities that manufacture or process dichlorvos, among professional exterminators or certified pesticide applicators involved in applying dichlorvos, and among workers at hazardous waste sites involved in the disposal of dichlorvos. Retained uses under the proposed EPA rule (September 28, 1995), would be restricted to trained, certified applicators, thereby eliminating most risks associated with application of this pesticide to all but occupational exposures (TOXLIST 1995). Dichlorvos is still widely used in agriculture, especially in confined animal breeding and feeding operations. Failure to wear protective clothing (especially gloves) or to rinse hands would increase the potential risks from dermal exposure for persons handling dichlorvos in concentrated liquid or resin-impregnated forms. Where dichlorvos is applied in concentrated liquid forms in interior spaces with poor ventilation (e.g., applications for termite control), inhalation risks could be significant without the use of protective clothing and equipment. Gold and Holcslaw (1985) evaluated dermal and inhalation exposure of pesticide applicators involved in spraying dichlorvos in domestic residences. Applicators wore a one-piece polyester jumpsuit with an open collar and long sleeves, a hard hat, respirator, and rubber gloves. The subject applicators were fitted with both inside and outside dermal exposure pads placed on the outer clothing and on the skin beneath the clothing. The total exposure calculated for the applicators was 0.0284 mg/kg/hour with 0.028 mg/kg/hour coming from dermal exposure and 0.0004 mg/kg/hour from respiratory exposure.

Information in the National Occupational Exposure Survey (NOES) conducted by NIOSH from 1981 to 1983 provides estimates of the number of workers and the different types of occupations and industries where workers could be potentially exposed to dichlorvos (NOES 1991). The NOES survey estimated 24,204 potential occupational exposures. Approximately 95% of these exposures are from four Standard Industrial Classification (SIC) codes: (1711) Plumbing, Heating and Air Conditioning; (2048) Prepared Feed Products; (2339) Women's and Misses' Outerwear; and (7342) Disinfecting and Exterminating Services.

The Occupational Safety and Health Administration (OSHA) has set a Permissible Exposure Limit (PEL) of 1.0 mg/m^3 (0.11 ppm) for dichlorvos in the workplace (OSHA 1974). The National Institute for Occupational Safety and Health (NIOSH 1992) also recommends an occupational exposure limit (time-weighted average [TWA]) of 1.0 mg/m^3 based on a 10-hour average workday and has designated a concentration exceeding 200 mg/m^3 as immediately dangerous to life and health (NIOSH 1990). The American Conference of Governmental Industrial Hygienists (ACGIH) recommends a Threshold Limit Value (TLV-TWA) of 0.90 mg/m^3 (skin) for occupational exposures (ACGIH 1994). The notation "skin" refers to the potential contribution to the overall exposure by the cutaneous route, including mucous membranes and the eyes, either by contact with vapors or by direct skin contact with dichlorvos. The use of this notation is intended to alert the reader that air sampling alone is insufficient to accurately quantitate exposure and that measures to prevent significant cutaneous absorption may be required (ACGIH 1994).

5.6 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

In addition to those individuals occupationally exposed to dichlorvos identified in Section 5.5, several groups within the general population may receive potentially higher inhalation exposures to dichlorvos. These groups include individuals living near facilities where dichlorvos is produced or processed and those individuals living near the 3 NPL hazardous waste sites where this compound is present. Although dichlorvos is not tightly bound to soil particles, ingestion of dichlorvos contaminated soil or soil where polyvinylchloride resin strips have been disposed of, may be a route of exposure particularly for children living in areas near production facilities or near hazardous waste sites. Ingestion of contaminated groundwater by individuals living in the vicinity of hazardous waste sites may be another possible source of exposure for both adults and children if these individuals use untreated well water as their primary source of drinking water.

5.7 ADEQUACY OF THE DATABASE

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

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

5.7.1 Identification of Data Needs

Physical and Chemical Properties. The relevant physical and chemical properties of dichlorvos (listed in Table 3-2), are generally known (ASTER 1996; Bowman and Sans 1983; Kenaga 1980; Kowamoto and Urano 1989; Merck 1989; Sunshine 1969; Worthing 1983). No major data needs were identified for physical and chemical properties.

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 chemical release and off-site transfer information to the EPA. The Toxics Release Inventory (TRI), which contains this information for 1993, became available in May of 1995. This database will be updated yearly and should provide a list of industrial production facilities and emissions.

As with most pesticide agents, it is virtually impossible to make quantitative estimates of dichlorvos production, use, disposal, and imported and exported volumes. While regulations govern the registration and labeling of pesticide agents, no national statistics are maintained on the amounts sold or actually used. For pesticide agents where production has shown major declines and there are few

remaining plants manufacturing the pesticide for domestic use, confidentiality considerations may prevent the public reporting of production data. Import and export statistics are generally maintained only for very broad categories of insecticides and not for specific pesticide agents. Under the federal Resource Conservation and Recovery Act (RCRA) system, pesticides are similarly covered under very broad waste categories that make it virtually impossible to track disposal patterns for specific pesticide agents. This presents some fundamental problems in making more than the most general sorts of exposure assessments. While some historic information is available on production and use (Cremlyn 1978; EPA 1976, 1987b, 1988a, 1991a, 1993a, 1994a; Gianessi 1986; IARC 1979, 1991; Marking 1992; Menz et al. 1974 in IARC 1991; PIP-Dichlorvos 1993; WHO 1989), current information for any of these categories (production, use, import/export, and disposal) is considered a major data need. This information would be helpful in evaluating potential exposures and risks to human health.

Environmental Fate. Empirical data on the environmental fate of dichlorvos are almost completely lacking. Because dichlorvos tends to undergo rapid hydrolysis (Howard 1991; PIP-Dichlorvos 1993) under typical environmental conditions, it has been assumed to degrade rapidly and to pose few risks to humans. Dichlorvos is an environmental degradation product of the pesticide trichlorfon and can also be metabolized in many plants from the pesticide naled (EPA 1988a; Menzie 1972; PIP-Naled 1994; PIP-Trichlorfon 1993). In addition, the pesticides trichlorfon and naled can decompose to dichlorvos during sample preparation and analysis. These two factors add further complications to interpreting detections of dichlorvos in environmental media. Since dichlorvos found at NPL or other waste sites may be in the form of discarded resin-plastic materials, further fate and transport research would be valuable in explaining how the dichlorvos impregnated in such plastics is actually released to air, water, or soils.

Bioavailability from Environmental Media. Dichlorvos can be absorbed following inhalation of contaminated workplace air, through dermal contact, and through consumption of contaminated food or water. Exposure to dichlorvos through inhalation or dermal contact directly with liquid formulations or with dichlorvos pest strips and residues on treated surfaces are probably the largest sources of exposure for the general population. Exposure to dichlorvos through ingestion of contaminated drinking water is expected to be insignificant since the compound is readily hydrolyzed in water, except at very low pH values (pH <4). Well water from areas around hazardous waste sites where dichlorvos has been detected should be monitored to determine whether exposures are insignificant. Exposure to dichlorvos through ingestion of contaminated foods also is expected to be insignificant

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since this pesticide is rarely found in raw foods at concentrations of concern and because routine food preparation procedures (i.e., washing and cooking) further reduce dichlorvos residues (IARC 1991; Nakamura et al. 1993). The lack of information on current levels of use makes precise exposure assessments extremely difficult. Since resin-strips are no longer used in households, risks to the general population are likely to have declined since the mid-1980s. Risks may still remain for persons in some occupations (e.g., professional pesticide applicators or workers in confined animal-feeding operations where dichlorvos is still used for pest control, and in veterinary applications) and for those people living near production or processing facilities or those living near waste disposal sites. The main occupational exposure pathways are from inhalation or dermal exposures. Although dichlorvos is not tightly bound to soil particles, dermal contact with or ingestion of dichlorvos contaminated soil or soil where polyvinylchloride resin strips have been disposed of, may be a route of exposure particularly for children living in areas near hazardous waste sites. Improvements in current estimates of the numbers of people in these various groups is a major data need. Information regarding the bioavailability of dichlorvos from dermal contact and from ingestion of soil-bound dichlorvos, particularly in children, and the levels of dichlorvos in contaminated groundwater would be helpful in assessing health risks to populations living near hazardous waste sites.

Food Chain Bioaccumulation. While virtually no experimental data could be identified, the BCF values calculated based on the physical and chemical properties of dichlorvos strongly suggest that there is very limited potential for bioaccumulation or biomagnification in the food chain (ASTER 1996; Kenaga 1980). Additional experimentally determined estimates of BCF in edible fish or shellfish would be helpful in verifying that bioaccumulation of dichlorvos is not an important process in the transfer of residues to humans.

Exposure Levels in Environmental Media. Dichlorvos is not commonly included in routine monitoring programs for air, water or soil. Additional monitoring data, at least in the form of national surveys such as the EPA's Pesticides in Groundwater project, would be very valuable in assessing ambient dichlorvos concentrations. Dichlorvos has been included in the FDA food surveillance program, and these data suggest there is no significant risk to the general U.S. population from eating dichlorvos in the food supply (FDA 1989, 1990, 1991, 1992, 1993, 1994c, 1995; KAN-DO Pesticide Team 1995). Available information also suggests there are few risks associated with ingestion of treated drinking water from larger community systems because water treatment procedures operate in alkaline conditions that would facilitate more rapid hydrolysis of dichlorvos. Exposures to dichlorvos

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from ingestion of contaminated groundwater from domestic wells are not well known. Additional monitoring data are needed to determine dichlorvos concentrations in domestic well water, especially from wells near waste disposal sites where dichlorvos has been identified and where the well water is being used as a drinking water source by local residents. This information is needed to determine whether any additional risks exist for populations living in proximity to these hazardous waste sites. Since dichlorvos found at NRL or other waste sites may be in the form of discarded resin-plastic materials, further research on the degradation of dichlorvos impregnated in such plastics would be useful in assessing exposure levels particularly in soil and sediment.

Exposure Levels in Humans. There are no data concerning levels of dichlorvos in human tissues (including adipose tissues, blood, milk, or urine) for members of the general U.S. population or in populations with occupational exposure to dichlorvos. Dichlorvos has not been detected in the human body and only rarely in animals because of its rapid metabolism by esterases in the liver and blood. Additional information on exposure levels in the general population and in individuals in occupational settings would be useful. This information is necessary for assessing the need to conduct health studies on these populations.

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

5.7.2 Ongoing Studies

The U.S. Department of Agriculture (USDA) is currently sponsoring several research projects related to migration of the pesticide trichlorfon following application to turfgrass (FEDRP 1996). This research is important in studying the movement of dichlorvos which is a major degradation product of trichlorfon. Researchers at the New York Agricultural Experimental Station in Geneva, New York are conducting experiments to determine the extent of downward movement of the pesticide trichlorfon commonly applied to creeping bentgrass turf grown on various soil types. Leachates and soils from

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the Automated Rain Exclusion System for Turfgrass Studies in Ithaca, New York will be analyzed for chemical residues.

Researchers at Cornell University in Ithaca, New York are conducting experiments to determine the extent of downward movement of the pesticide trichlorfon applied to several major soil types using normal management practices. The leaching characteristics of trichlorfon will be studied on creeping bentgrass turf plots. Leachates and soils from the Automated Rain Exclusion System for Turfgrass Studies in Ithaca, New York will be analyzed for chemical residues.

Researchers at Pennsylvania State University in University Park, Pennsylvania are conducting experiments to determine the extent of downward movement of the pesticide trichlorfon applied to several major soil types using normal management practices. The runoff and leaching characteristics of trichlorfon will be studied on creeping bentgrass turf plots. Leachates and soils from the Automated Rain Exclusion System for Turfgrass Studies in Ithaca, New York will be analyzed for chemical residues.

Researcher at the University of Massachusetts at Amherst, Massachusetts are conducting experiments to determine the rate of volatile loss and the amount of dislodgeable residues from pesticide-treated turfgrass. One of the pesticides being studied is trichlorfon. The fate of the pesticide will be evaluated. Concentrations of trichlorfon will be measured in air samples and in dislodgeable residues from a one square foot area of treated sod. Preliminary investigations have found that dichlorvos concentrations in air may be of concern even three days post-application of trichlorfon on the grass.