

6. POTENTIAL FOR HUMAN EXPOSURE

6.1 OVERVIEW

Aldrin was first synthesized in the United States as a pesticide in 1948 (EPA 1986d) while dieldrin was first used by cotton growers in the 1950s (Clayton and Clayton 1994). By 1970, the U.S. Department of Agriculture canceled all uses of aldrin and dieldrin (EPA 1980a). Restrictions on their use as termiticides, for dipping of non-food plant roots and tops, and for moth-proofing were lifted by EPA in 1972. In 1974, however, the latter two uses were voluntarily canceled by the manufacturer, Shell Chemical Company (EPA 1986d). The final registered use of aldrin and dieldrin as termiticides was voluntarily canceled by the Scallop Corporation (part of the Shell Chemical Company) on May 15, 1987 (EPA 1989a). The Chapman Chemical Company, however, continued to use aldrin in their termiticide formulation until it was ultimately canceled by the EPA on February 21, 1989.

Aldrin is readily converted to dieldrin, which is ubiquitous in the environment. Dieldrin persists because it is more resistant to biotransformation and abiotic degradation than aldrin. As a result, it is found in all environmental media, even at a distance from the site of concentration. Dieldrin bioconcentrates and biomagnifies through the terrestrial and aquatic food chains. Transport of aldrin and dieldrin in soils is minimal because these compounds tend to bind tightly to soil. Based on their physical properties, volatilization from moist soil surfaces is expected. Most dieldrin and aldrin found in surface water are the result of runoff from contaminated soil. Aldrin undergoes photolysis to dieldrin, which in turn may be degraded by ultraviolet radiation or microbial action into the more persistent compound, photo dieldrin.

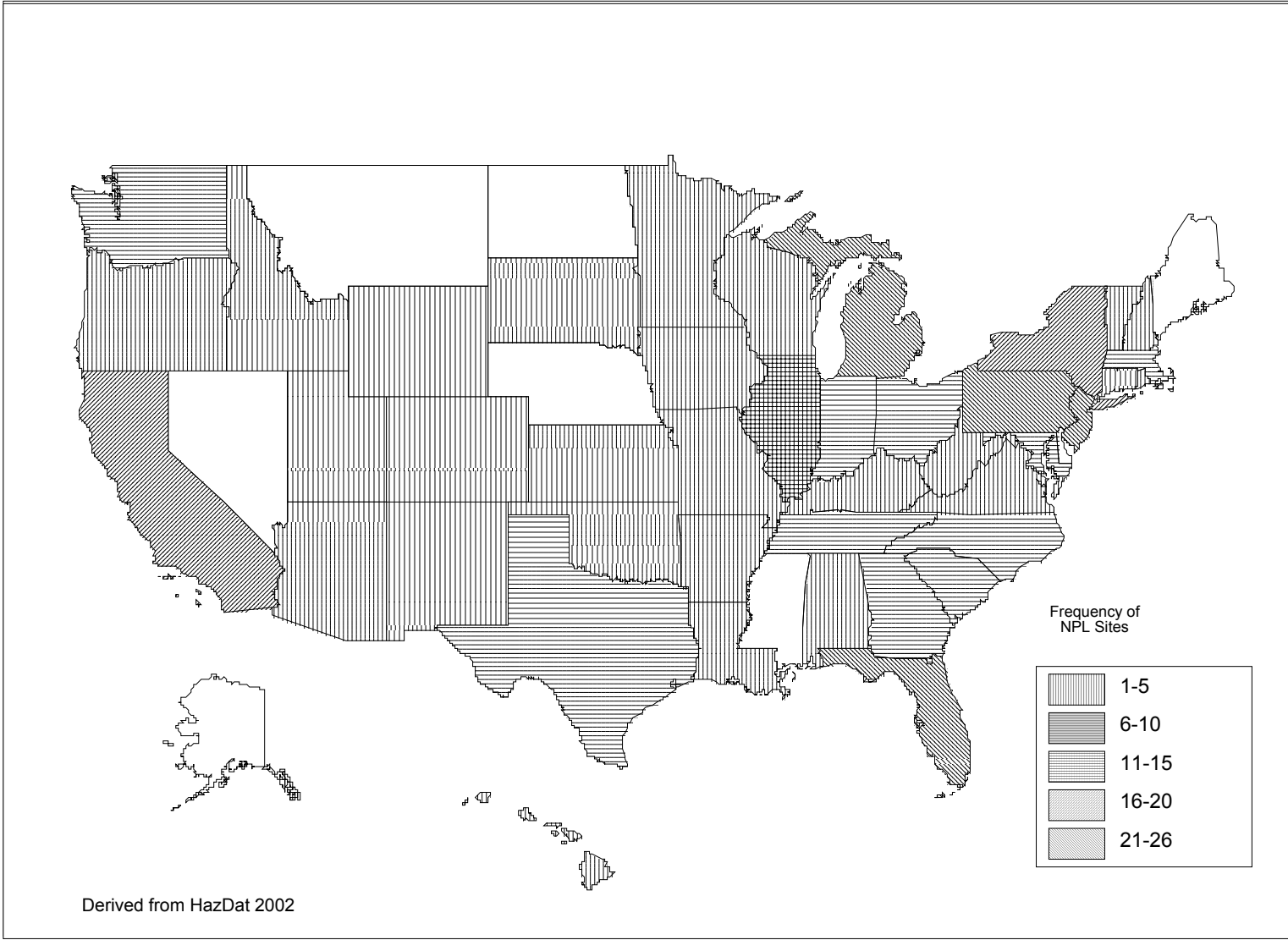
Past agricultural uses of aldrin and dieldrin have resulted in persisting soil residues and uptake in a wide range of crops. Exposure of the general population to aldrin and dieldrin may occur through ingestion of contaminated water or food products and through inhalation of contaminated air, especially in homes that have been treated with either pesticide.

Aldrin has been identified in at least 207 of the 1,613 hazardous waste sites while dieldrin has been identified in at least 287 of the 1,613 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HazDat 2002). However, the number of sites evaluated for aldrin and dieldrin is not known. The frequency of these sites can be seen in Figures 6-1 and 6-2. Of these

Figure 6-1. Frequency of NPL Sites with Aldrin Contamination



Figure 6-2. Frequency of NPL Sites with Dieldrin Contamination



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sites, 205 of the 207 aldrin sites are located within the United States, 1 is located in the Virgin Islands (not shown), and 1 is located in Puerto Rico (not shown). For dieldrin, 285 of the 287 sites are located within the United States and 2 are located in the country of Guam.

6.2 RELEASES TO THE ENVIRONMENT

Aldrin and dieldrin production and use in the United States has been canceled by the EPA (EPA 1990b). Because of the persistent nature of these compounds, however, these compounds are still present in the environment. Aldrin and dieldrin have been identified in a variety of environmental media (air, surface water, groundwater, soil, and sediment) collected at 207 and 287 of the 1,613 NPL hazardous waste sites, respectively (HazDat 2002).

Aldrin and dieldrin are included on the most recent Toxic Chemical Release Inventory (TRI99) as reportable chemicals when released or transferred from TRI99 facilities (TRI99 2001). EPA received one TRI99 form from Safety Keen (Deer Park) Inc. located in Deer Park, Texas. This facility performed the waste treatment of aldrin and reported no releases to the environment. EPA received no other release data for aldrin or dieldrin, indicating that no reportable releases to the environment occurred in 1999.

The TRI data should be used with caution because only certain types of facilities are required to report. This is not an exhaustive list.

6.2.1 Air

Aldrin has been identified in air samples collected at 6 of the 207 NPL hazardous waste sites where it was detected in some environmental media (HazDat 2002). Dieldrin has been identified in air samples collected at 14 of the 287 NPL hazardous waste sites where it was detected in some environmental media (HazDat 2002).

Past application of aldrin and dieldrin for termite control is a continuing source of contamination of indoor air. In addition, these compounds may be released to the atmosphere from previously treated soil and contaminated surface waters. Release of aldrin and dieldrin into the air may also occur as a result of atmospheric dispersal of contaminated soils at NPL sites and farmlands where these compounds had been used.

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6.2.2 Water

Aldrin has been identified in 20 surface water and 93 groundwater samples collected at 207 of the 1,613 NPL hazardous waste sites where it was detected in some environmental media (HazDat 2002). Dieldrin has been identified in 40 surface water and 107 groundwater samples collected at 287 of the 1,613 NPL hazardous waste sites where it was detected in some environmental media (HazDat 2002).

Aldrin and dieldrin may be released to surface waters as a result of runoff from contaminated croplands and NPL sites. Although aldrin and dieldrin are no longer permitted for general use, dieldrin, in particular, has been detected in many waterways and cropping soils. Due to the persistence of these compounds, especially dieldrin, they have been detected in a wide variety of aquatic systems. Aldrin and dieldrin have been detected in seawater samples (Sauer et al. 1989), industrial effluents and fresh water samples (Staples et al. 1985). The high organic carbon partition coefficient (K_{oc}) values for aldrin and dieldrin suggest movement through soil and contamination of groundwater will be minimal. The only reports of aldrin or dieldrin contamination of groundwater occurred at sites with high concentrations of these compounds.

6.2.3 Soil

Aldrin has been identified in 145 soil and 45 sediment samples collected at 207 of the 1,613 NPL hazardous waste sites where it was detected in some environmental media (HazDat 2002). Dieldrin has been identified in 243 soil and 89 sediment samples collected at 287 of the 1,613 NPL hazardous waste sites where it was detected in some environmental media (HazDat 2002).

Possible releases of aldrin and dieldrin to soil may come from the improper disposal of old stocks. Wet and dry deposition of particulate phase aldrin and dieldrin from the atmosphere is another potential source of soil contamination.

6.3 ENVIRONMENTAL FATE

6.3.1 Transport and Partitioning

Experimental log K_{oc} values for aldrin range from 5.38 to 7.67 (Briggs 1981; Ding and Wu 1995). Based on a classification scheme, these log K_{oc} values indicate that aldrin is expected to be immobile in soil (Swann et al. 1983). The mobility of aldrin and dieldrin in the soil environment, however, can be

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enhanced at hazardous waste sites where organic solvents may be present. These organic solvents have the ability to increase the water solubility of nonpolar compounds which in turn increases their mobility in soil (Sawhney 1989). The organic solvents in a sense act as a transport medium for chemicals that would normally bind strongly to soil. At waste disposal sites, where bioremediation techniques are proposed to reduce the mass of carbon-containing contaminants, there is the potential for augmenting the leaching properties of organochlorine compounds such as aldrin and dieldrin. The lipid materials in bacterial cell membranes may lead to a repartitioning of aldrin and dieldrin sorbed to soil colloids. This can lead to a phenomenon called facilitated transport where the mobility of hydrophobic pollutants adsorbed to soils may be enhanced by biosorption on bacteria and move into aquifers along with the bioremedial bacterial cultures (Lindqvist and Enfield 1992). Except at NPL sites, however, this potential source of groundwater pollution would seem to be remote. This appears to be true in light of the small number of reports of aldrin and dieldrin groundwater contamination at locations other than NPL sites. Volatilization of aldrin from soil is more rapid when it is applied to the soil surface rather than incorporated into the soil. A loss of 50% from a surface application was estimated to occur within 1–2 weeks after application compared to 10–15 weeks for soil-incorporated aldrin (Caro and Taylor 1971; Elgar 1975). The relatively rapid loss of both aldrin and dieldrin from soil during the first few months after application has been attributed to loss by volatilization. The volatilization potential of field-applied dieldrin (10 ppm) was studied for 5 months using three different soil moisture regimes (Willis et al. 1972). The three soil moisture regimes included: (1) flooded to a depth of 10 cm; (2) moist; and (3) nonflooded with no water added except for natural rainfall. The results showed that the soil moisture had an effect on the volatilization rate. About 18% of the applied dieldrin volatilized from a moist plot in 5 months, but only 2 and 7% volatilized from the flooded and nonflooded plots, respectively. Flooding retarded the volatilization potential of surface-applied dieldrin. Volatilization of dieldrin from the non-flooded plot tended to increase with increasing precipitation (Willis et al. 1972).

Volatilization of aldrin from water surfaces is expected (Thomas 1990) based upon a Henry's law constant of 4.9×10^{-5} atm/m³/mole (Guerin and Kennedy 1992). Volatilization from water surfaces, however, may be attenuated by adsorption to suspended solids and sediment in the water column. The volatile loss of aldrin from sterile, deionized water kept at 30 EC was studied over a 30 day period (Guerin and Kennedy 1992); the volatilization half-life of aldrin from the open flask was 5.8 days. In one study, the desorption of aldrin from sediment into water was investigated (Ding and Wu 1993). Researchers simulated a river bed by spiking a sediment sample with 287.6 ng aldrin/g sediment and passing 7,780 mL water/day over the sediment. The concentration of aldrin detected in the effluent water after one day was 0.135 ng/mL.

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while by day 40, the concentration decreased to 0.06 ng/mL. The concentration of dieldrin was not measured in this study.

According to a model of gas/particle partitioning of semivolatile organic compounds in the atmosphere (Bidleman 1988), aldrin, which has a vapor pressure of 1.2×10^{-4} mmHg at 25 EC (HSDB 2001a), will exist in both the vapor and particulate phases in the ambient atmosphere. Particulate-phase aldrin may be transported through the atmosphere by wind and later removed from air by wet and dry deposition (Millet et al. 1997).

The logarithm of the *n*-octanol/water partition coefficient ($\log K_{ow}$) is a useful preliminary indicator of potential bioaccumulation of a compound. The $\log K_{ow}$ for aldrin ranges from 5.68 (McLean et al. 1988) to 7.4 (Briggs 1981), indicating a high potential for bioaccumulation. In modeling ecosystem tests, bioconcentration factors (BCFs) for aldrin were 3,140 in fish and 44,600 in snails (Metcalf et al. 1973). The BCF of aldrin in orange-red killifish was studied over an 8 week period in a semi-static system at 25 EC (CITI 1992.). At a concentration of 1 mg/L, aldrin had BCFs ranging from 3,490 to 20,000, while at 0.1 mg/L, aldrin had BCFs ranging from 1,550 to 9,450.

Experimental evidence indicates that aldrin is rapidly metabolized to dieldrin by some organisms, which then bioconcentrates and biomagnifies (EPA 1980a; Metcalf et al. 1973). Radiolabeled aldrin added to a model ecosystem was rapidly converted to dieldrin. Of the radiolabel stored in organisms, 95.9% of the total stored in the fish *Gambusia affinis*, 91.6% stored in the snails of the genus *Physa*, and 85.7% stored in the algae *Oedogonium cardiacum* were in the form of dieldrin.

Aldrin also bioconcentrates in terrestrial ecosystems. In a model ecosystem study, 2.09 ppm radiolabeled aldrin was applied to a vermicullite soil (Cole et al. 1976). After 20 days, researchers detected only 0.463 ppm aldrin and 0.159 ppm dieldrin. Corn, that had been grown on the vermicullite soil for 14 days, contained 2.83 ppm radiolabeled carbon with 0.762 ppm being aldrin and 1.538 ppm dieldrin.

Approximately 78% of the plant residue was in the roots and 22% in the shoots. On day 15, a prairie vole (*Microtus ochrogaster*) was introduced to the model ecosystem. After 5 days of exposure, the concentrations of aldrin and dieldrin in the vole were 0.08 and 3.56 ppm, respectively. To study the uptake of pesticides in plants, radiolabeled aldrin and dieldrin were monitored over one week in a controlled laboratory setting (Kloskowski et al. 1981). After 1 week of exposure of barley plants to 2 ppm of both pesticides, the concentrations of aldrin and dieldrin in plant tissue were 9.7 and 4.0 ppm, respectively. One research study, however, observed no plant uptake of either aldrin or dieldrin in maize

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and pearl millet over a 3-year period grown in a clay loam soil (Gupta et al. 1979). Aldrin was applied at a rate of 3, 9, and 15 kg active ingredient (ai) per hectare (ha) once per year before the sowing of crops and mixed up to a depth of 10 cm. No residues of either aldrin or dieldrin could be detected in plant tissues from any of the years of experimentation, even at the highest dosage of 15 kg ai/ha.

Biotransfer factors (BTFs) for beef and cow's milk have been determined for aldrin. The concept of biotransfer is useful since it takes into account exposure through both food and water pathways. Biotransfer factors for beef and milk are defined as the concentration of a compound in beef or milk (mg/kg) divided by the daily intake of the compound by the animal (mg/day). The biotransfer values for beef and milk were estimated to be 0.085 and 0.023, respectively (Travis and Arms 1988). Biotransfer factors for aldrin in beef and milk are directly proportional to the K_{ow} . In addition, a BCF for aldrin in vegetables was also determined. The bioconcentration factor was defined as the ratio of the concentration in aboveground parts (mg of compound/kg of dry plant) to the concentration in soil (mg of compound/kg of dry soil); the BCF was estimated to be 0.021 (Travis and Arms 1988). The vegetation bioconcentration factor is inversely proportional to the square root of K_{ow} . The regression equations for beef, milk, and vegetation provide a technique for predicting a chemical's BTF in beef and milk and BCF in vegetation. Consequently, regression analyses will be of value in more precisely quantifying human exposure to organics through the terrestrial food chains (Travis and Arms 1988).

Dieldrin is nonpolar and, therefore, has a strong affinity for organic matter and sorbs tightly to soil particulates based on its log K_{oc} of 6.7 (Briggs 1981). Volatilization is the principal loss process of dieldrin from soil; however, the process is relatively slow due to its low vapor pressure and strong sorption to soil. It may also be impeded by low soil moisture or incorporation of the compound into the soil (Clith and Spencer 1971). Volatilization of dieldrin from dry soil is slower than aldrin (<10 g/hectare/day) based on its vapor pressure of 5.89×10^{-6} mmHg at 25 EC (Grayson and Fosbraey 1982). The volatilization rate decreases with time (Nash 1983) and increases with increasing temperature to a maximum at 25 EC (Nash and Gish 1989). In one experiment, 150 grams of dieldrin was applied to a sandy loam soil and monitored for volatile loss (Nash 1983). After 11 days, 53.1 ± 14.2 mg had volatilized, while 110 ± 18.6 mg had remained on the soil surface. Based on the Henry's law constant and the K_{oc} , the volatilization half-life of dieldrin from soil has been estimated to be 868 days (Jury et al. 1987b). Movement of dieldrin through the soil solution is extremely slow, indicating little potential for groundwater contamination. Using a low pollution potential scenario (soil with a high organic content and a high average water content), it is estimated that it will take 2,594 years for dieldrin to travel to a depth of 3 meters. Even with a high pollution potential scenario (soil with a low organic content and a

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low water content), it would still take an estimated 270 years for dieldrin to reach a depth of 3 meters (Jury et al. 1987b). Analysis of environmental groundwater samples, however, have shown that on some occasions, dieldrin has contaminated groundwater systems (EPA 1986i; Hallberg 1989; HazDat 2001). Dieldrin has been estimated to have a sorption coefficient on mixed-liquor solids (typical of municipal waste water treatment plants) of 38.9 mg/g at an equilibrium concentration of 1.0 mg/L (Dobbs et al. 1989). Movement of dieldrin in waterborne sediment is a major loss pathway from treated soil (Caro and Taylor 1971; Eye 1968; Hardee et al. 1964).

Volatilization of dieldrin from water surfaces is expected (Thomas 1990) based upon a Henry's law constant of 5.2×10^{-6} atm/m³/mole (Guerin and Kennedy 1992). Volatilization from water surfaces, however, may be attenuated by adsorption to suspended solids and sediment in the water column. The volatile loss of dieldrin from sterile, deionized water kept at 30 EC was studied over a 30-day period (Guerin and Kennedy 1992). The study found that the half-life for the volatilization of dieldrin from the open flask was 17 days.

According to a model of gas/particle partitioning of semivolatile organic compounds in the atmosphere (Bidleman 1988), dieldrin, which has a vapor pressure of 5.89×10^{-6} mmHg at 25 EC (Grayson and Fosbraey 1982), will exist in both the vapor and particulate phases in the ambient atmosphere. Dieldrin may be transported great distances in the atmosphere and be removed by wet or dry deposition (Baldwin et al. 1977; Millet et al. 1997). Snowpack samples were collected at 12 sites in the Northwest Territories, Canada, in the winter of 1985–1986; dieldrin was found in all 21 samples at a mean concentration of 0.75 ng/L (Gregor and Gummer 1989). There were no known local sources of dieldrin in the Canadian Arctic snow. Dieldrin was detected with a mean concentration close to 1 pg/m³ in arctic air measured at Alert, Canada; Tagish, Canada; and Dunai Island, Canada-Russia in the 1990s (Bidleman 1999).

Like aldrin, dieldrin has a high potential for bioaccumulation as indicated by a log K_{ow} value that ranges from 4.32 (Geyer et al. 1987) to 6.2 (Briggs 1981). Measured bioconcentration factors for dieldrin are 2,700 in fish and 61,657 in snails (Metcalf et al. 1973). A second study using the same model ecosystem found bioconcentration factors for dieldrin to be 6,145 in fish, 7,480 in algae, 247 in crabs (*Uca minax*), 1,015 in clams (*Corbicula manilensis*), 1,280 in the water plant *Elodea*, and 114,935 in snails (Sanborn and Yu 1973). A BCF of 2,095 has been determined for the ciliate *Tetrahymena pyriformis* exposed to 1 µg/mL dieldrin for 12 hours (Bhatnagar et al. 1988). A biomagnification factor of 1.0 has been determined for dieldrin for rainbow trout on a lipid weight basis; the average wet weight bioconcentration factor is 2.3 (Connell 1989). Channel catfish, exposed to varying concentrations of dieldrin, were used to

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determine when equilibrium was reached between uptake of dieldrin and elimination from muscle tissue. At 13 ppt, equilibrium was reached after 56 days, whereas, at 49 ppt, equilibrium was not reached even after 70 days of exposure (Shannon 1977). The bioaccumulation factor of dieldrin in orange-red killifish was studied over an 8-week period in a semi static system at 25 EC (CITI 1992). At a concentration of 1 mg/L, dieldrin had BCFs ranging from 4,860 to 14,500 while at 0.1 mg/L, dieldrin had a BCF ranging from 5,390 to 12,500.

Biotransfer factors for beef and cow milk and a bioconcentration factor for vegetables have been determined for dieldrin. The biotransfer values for dieldrin in beef and milk were estimated to be 0.008 and 0.011, respectively, while the BCF for vegetables was estimated to be 0.098 (Travis and Arms 1988).

In a biomagnification study, the concentrations of organochlorine compounds in sediments, amphipods, isopods, and sculpins from the Bothnian Bay and the Bothnian Sea were measured (Strandberg et al. 2000). Dieldrin was detected in sediments (three samples), amphipods (three samples), isopods (five samples), and sculpins (three samples) in the Bothnian Bay with mean concentrations of 0.39, 87, 92, and 42 ng/g lipid, respectively. Dieldrin was detected in sediments (three samples), amphipods (four samples), isopods (five samples), and sculpins (three samples) with mean concentrations of 0.51, 110, 55, and 80 ng/g lipid, respectively. Possible explanations given for the low biomagnification factor potential of the sculpin were that it could have less capacity to accumulate hydrophobic organic environmental contaminants or a greater ability to metabolize or excrete the compounds.

Uptake of dieldrin by redworms (*Eisenia foetida*) was determined for Chester and silt loam samples that had been aged with dieldrin for periods of 49 and 30 years, respectively (Morrison et al. 2000). The worms assimilated 10.8% of the dieldrin in unaged Chester loam resulting in a tissue concentration of 53.5 mg/kg. The worms assimilated 4.48% of the dieldrin in the Chester loam aged 49 years resulting in a tissue concentration of 15.1 mg/kg. In unaged silt loam, the worms assimilated 12.8% of the dieldrin resulting in a tissue concentration of 40.0 mg/kg, while the worms in the silt loam aged 30 years assimilated 19.9% of the dieldrin resulting in a tissue concentration of 6.13 mg/kg. It was suggested that the aging dieldrin in field soils reduced acute toxicity and therefore bioavailability to earthworms.

Data indicate that dieldrin is taken up by various crops (Beall and Nash 1969, 1971). To determine whether foliar contamination of soybean plants occurred via root sorption or vapor sorption, 20 ppm ¹⁴C-dieldrin was applied to surface or subsurface soil, and residue levels in soybean plants were

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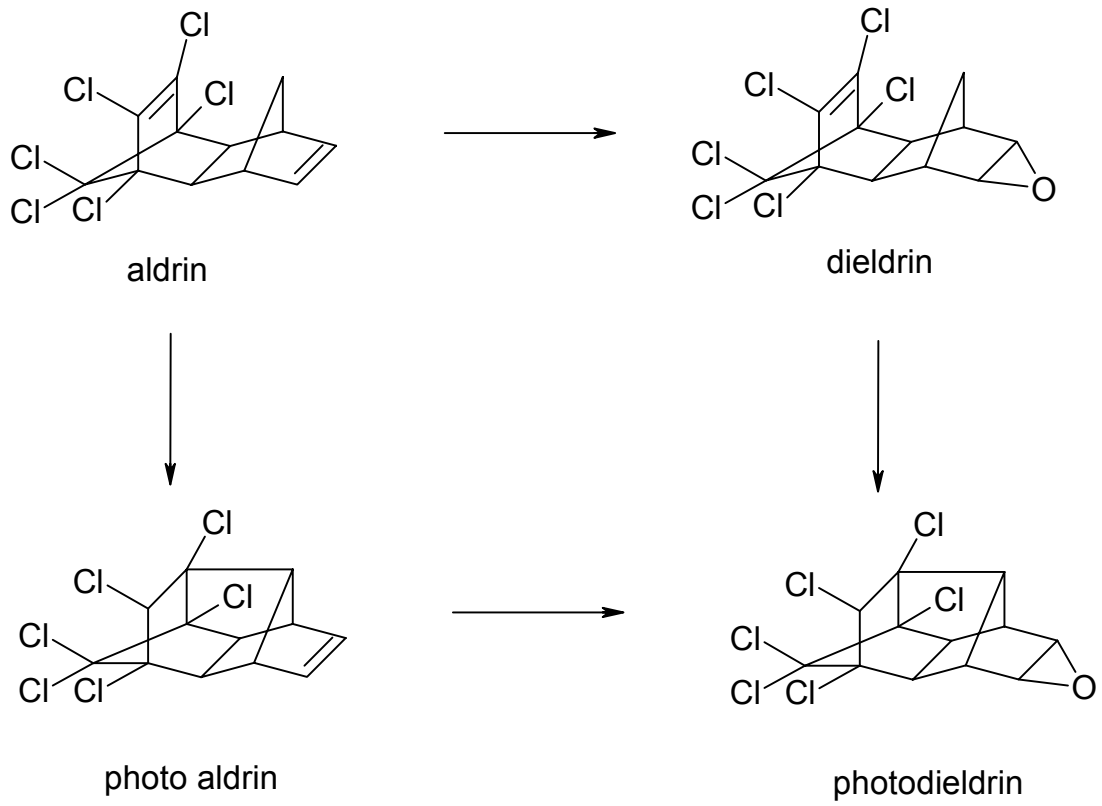
determined (Beall and Nash 1971). The results indicated that foliar contamination by dieldrin occurred by both root sorption (10.8 ppm) and vapor sorption (8.5 ppm) (Beall and Nash 1971). In a greenhouse experiment, various crop seedlings took up dieldrin from soils treated with 0.5 or 5.0 ppm dieldrin (Beall and Nash 1969). Mean concentrations of dieldrin found in soybeans, wheat, corn, alfalfa, brome grass, and cucumber treated with 0.5 ppm dieldrin were 0.017, 0.147, 0.017, 0.031, 0.075, and 0.070 ppm (dry weight). Mean concentrations of dieldrin found in soybeans, wheat, corn, alfalfa, brome grass, and cucumber treated with 5.0 ppm dieldrin were 0.194, 1.385, 0.171, 0.350, 0.808, and 0.185 ppm (dry weight) (Beall and Nash 1969).

6.3.2 Transformation and Degradation

6.3.2.1 Air

While the evidence supports the view that a considerable proportion of the aldrin and dieldrin used in agriculture reaches the atmosphere, it seems probable that atmospheric degradation and wet and dry deposition prevents accumulation of aldrin. In laboratory studies, vapor-phase aldrin is photochemically isomerized and epoxidated by sunlight to photoaldrin, dieldrin, or photodieldrin (see Figure 6-3) (Glotfelty 1978). In order to determine the potential for photodegradation to occur in the ambient atmosphere, the degradation of aldrin and dieldrin was studied on thin film plates and exposed to environmental ultraviolet (UV) radiation (>290 nm) (Chen et al. 1984). Aldrin and dieldrin had photodegradative half-lives of 113 and 153 hours, respectively. Researchers also reported that aldrin and dieldrin have UV absorbance maximums of 227 and 229 nm, respectively. Irradiation of aldrin (5 mg) vapor with ultraviolet light for 45 hours resulted in the formation of photoaldrin (20–30 ug) and dieldrin (50–60 ug). Irradiation of either photoaldrin (2 mg) or dieldrin (0.5 mg) vapor for 65 hours and 91 minutes, respectively, resulted in a single photoproduct, photodieldrin (20–30 ug), which was resistant to further photolyses (Crosby and Moilanen 1974). Since photodieldrin no longer contains a chromophore, it is believed to be a stable photoproduct of aldrin (dieldrin) (Glotfelty 1978). Results of a

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Figure 6-3. Aldrin Degradation

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laboratory study, however, revealed that photolysis of photoaldrin and photodieldrin in the presence of triethylamine gave photometabolites arising from the loss of chlorine atoms (Dureja et al. 1986).

Information regarding the persistence of photodieldrin in the atmosphere was not located; however, air samples taken in 1973 in Ireland contained dieldrin, but neither aldrin nor the photoproducts of aldrin or dieldrin were detected (Baldwin et al. 1977).

Vapor-phase aldrin and dieldrin are expected to degrade in the atmosphere by reaction with photochemically-produced hydroxyl radicals. The half-lives for this reaction in air are estimated to range from 1 to 10 hours for aldrin and 3 to 30 days for dieldrin based on an estimated rate constant (Kwok and Atkinson 1995). Vapor-phase aldrin may also be degraded in the atmosphere by reaction with ozone. Although there are no experimental data, reaction with ozone is expected to be an important atmospheric degradation reaction for aldrin in the vapor phase. An estimated half-life for this reaction ranges from 19 minutes to 2 hours (Atkinson and Carter 1984). Studies indicate that aldrin will also react with nitrogen dioxide in the ambient atmosphere to produce dieldrin (Nojima et al. 1982). After 3 hours of exposure to nitrogen dioxide and UV radiation >290 nm, 32% of vapor-phase aldrin was converted to dieldrin. Aldrin and dieldrin may be more stable than implied by these lifetimes if they are associated with particulate matter in the atmosphere. Particulate-phase aldrin and dieldrin, however, will not participate in hydroxyl radical reactions in the atmosphere.

6.3.2.2 Water

The resistance of aldrin and dieldrin to soil leaching generally precludes their appearance in groundwater. The general absence of aldrin and dieldrin from groundwater samples supports this conclusion (Richard et al. 1975; Spalding et al. 1980). The potential for surface runoff of aldrin and dieldrin in soils is supported by reports of detectable quantities of these compounds in surface waters (Hindin et al. 1964; Richard et al. 1975).

Aldrin, irradiated with ultraviolet light in an oxygenated aqueous solution, underwent little change except in the presence of amino acids and humic acids present in natural waters (Ross and Crosby 1975, 1985). In filtered natural field water, aldrin was photooxidized by 75% to dieldrin after 48 hours of irradiation at 238 nm (Ross and Crosby 1985). More than 80% of the initial dieldrin added to natural water (from a drainage canal in an agricultural area) was present after 15 weeks of incubation in the dark (Sharom et al. 1980). Dieldrin exposed to sunlight is converted to photodieldrin, a stereoisomer of dieldrin. It is unlikely, however, that photodieldrin occurs widely in the environment. Microorganisms isolated from

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lake water and lake-bottom sediments may convert dieldrin to photodieldrin under anaerobic conditions (Fries 1972). The stability of dieldrin and aldrin was determined in distilled (pH 6.8) and roof water (pH 7.4) (McDougall et al. 1994). The samples were kept in the dark, at 23 EC over a 36-week period. The study found that after 36 weeks, dieldrin remained stable while aldrin degraded in both roof water and distilled water. The half-lives of aldrin in distilled water and roof water were 4.9 and 5.1 weeks, respectively. The study did not find dieldrin as a breakdown product of aldrin degradation. An extrapolated hydrolysis rate constant of $3.8 \times 10^{-5} \text{ hour}^{-1}$ at pH 7 and 25 EC has been determined for aldrin based on a measured value at 75 EC (EPA 1989d). The half-life for this reaction is 760 days.

Aldrin was degraded under anaerobic conditions in biologically active waste water sludge (pH 7–8, 35 EC) with a half-life of <1 week (Hill and McCarty 1967). Under aerobic conditions, however, only 1.5% of aldrin degraded when exposed to an activated sewage sludge (Freitag et al. 1985). Aldrin has a reported biodegradation half-life of 24 days in surface waters based on a non-acclimated river die-away test (Eichelberger and Lichtenberg 1971). Dieldrin does not undergo any significant degradation in biologically active waste water sludge or by sewage sludge microorganisms under anaerobic conditions (Battersby and Wilson 1988; Hill and McCarty 1967). After 48 hours of continuous anaerobic digestion with primary sludge, dieldrin was degraded by only 11% (Buisson et al. 1990). Likewise, when incubated for 32 days with anaerobic sludge, only 24% of the dieldrin was removed (Kirk and Lester 1988). In contrast, aerobic incubation with activated sludge removed 55% of the dieldrin in 9 days (Kirk and Lester 1988). A mixed, anaerobic microbial enrichment culture was able to degrade 10 µg/mL dieldrin by 50% in 30 days. Syn-monodechlorodieldrin and anti-monodechlorodieldrin, both of which are resistant to microbial degradation, were identified as the initial degradation products (Maule et al. 1987). In another study, dieldrin was degraded by 30–60% using activated sludge treatment, with the most effective removal by activated sludge aged 4 days as opposed to sludge aged 6 and 9 days (Buisson et al. 1988). Both aldrin and dieldrin, present at 100 mg/L, reached 0% of their theoretical biological oxygen demand (BOD) in 2.5 weeks using an activated sludge inoculum at 30 mg/L and the Japanese Ministry of International Trade and Industry (MITI) test (CITI 1992).

Dieldrin undergoes minor degradation to photodieldrin in marine environments. The marine algae of the genus *Dunaliella* had the maximum degradation activity, degrading 23% of aldrin to dieldrin and 8.5% of dieldrin to photodieldrin (Patil et al. 1972).

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6.3.2.3 Sediment and Soil

In the soil, aldrin is converted to dieldrin by epoxidation (Gannon and Bigger 1958). Aldrin epoxidation occurs in all aerobic and biologically active soils, with 50–75% of end-season residues detected as dieldrin. The transformation of aldrin to aldrin acid also occurs in soils. The half-life of aldrin in soil is estimated to be 53 days. Mathematical modeling estimates that aldrin, applied to soil up to 15 cm in depth, will degrade to dieldrin by 69% after 81 days. At a typical soil application rate of 1.1–3.4 kg/hectare, the half-life of aldrin was estimated to be 0.3 years with 95% disappearance in 3 years (Freedman 1989). Loam soils treated with aldrin at 25 pounds per 5-inch acre over a 5-year period from 1958 to 1962 contained in the fall of 1968, 4–5% of the applied dosages mainly in the form of dieldrin. Aldrin treated soils also contained photodieldrin, which amounted to 1.5% of the recovered dieldrin (Lichtenstein et al. 1970). The degradation of aldrin and dieldrin was studied under upland and flooded soil conditions (Castro and Yoshida 1971). For the upland soil condition, water was added to give 80% of the maximum water-holding capacity of the soil. For the flooded soil condition, the water level was maintained 5 cm above the soil surface resulting in an anaerobic environment. Results showed that aldrin was more persistent in flooded than in upland soil. After 2 months of incubation under upland conditions, 33–58% of added aldrin remained in the soil. Under flooded conditions, 64–81% remained in the soil (Castro and Yoshida 1971).

The change in aldrin concentration and its conversion to dieldrin was also studied over a 3-year period in a clay loam soil in India (Gupta et al. 1979). Aldrin was applied at a rate of 3, 9, and 15 kg active ai/ha once per year before the sowing of crops and mixed up to a depth of 10 cm. After the first year of application at 3, 9, and 15 kg ai/ha, the concentrations of aldrin in soil were 1.801, 3.665, and 8.797 ppm, respectively. By the end of the third year, the concentrations of aldrin was 1.824, 3.453, and 9.736 ppm for the three application rates of 3, 9, and 15 kg ai/ha, respectively. Dieldrin was detected as a breakdown product of aldrin by the third year at a concentration of 0.055, 0.245, and 0.695 ppm for the three application rates, respectively. Maize and pearl millet grown on the treated soil were also analyzed for aldrin and dieldrin concentrations. No residues of either aldrin or dieldrin could be detected in plant samples from any of the years of experimentation, even at the highest application rate of 15 kg ai/ha.

Dieldrin is much more resistant to biodegradation than aldrin (Castro and Yoshida 1971; Gannon and Bigger 1958; Jagnow and Haider 1972; Willis et al. 1972). Of 20 soil microbes that were able to degrade dieldrin, only 13 of them could also degrade aldrin to dieldrin (Patil et al. 1970). The bacteria *Aerobacter aerogenes* aerobically degraded approximately 12% of dieldrin to aldrin diol within 5 days, but no further

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degradation was detected with increased incubation periods (Wedemeyer 1968). At a soil application rate of 1.1–3.4 kg/hectare, dieldrin was estimated to have a half-life of 2.5 years and a 95% disappearance from soil in 8 years (Freedman 1989), although other studies indicate that dieldrin loses between 75 and 100% of its biological activity in 3 years (Jury et al. 1987b). After 6 months, dieldrin persisted in moist, flooded, and nonflooded soils, indicating that these three soil moisture conditions had no effect on the degradation of soil-incorporated dieldrin (Willis et al. 1972). The roots of grass grown on the plots contained 11.6 ppm dieldrin while the aerial grass parts contained only 0.05 ppm (Voerman and Besemer 1975). Twenty-one years after the application of dieldrin to the foundation of a house at an application rate commonly used for termite control, 10% of the original dieldrin remained, primarily in the upper 6 inches of soil (Bennett et al. 1974). Aldrin and dieldrin applied to soil may also undergo degradation by ultraviolet light to form photodieldrin; this reaction may occur as a result of microbial action as well (Matsumura et al. 1970; Suzuki et al. 1974). After ultraviolet irradiation for 168 hours, dieldrin applied to various environmental media was found to be photodecomposed by 9.6% on loam soil, 1.2% on clay soil, and 44% on activated charcoal; the degradation products were photodieldrin and an unknown compound (Elbeit et al. 1983). Residues in soil samples found after application of dieldrin to soil (0.83 kg/hectare in soil that already contained 0.521 ppm dieldrin) consisted largely of unchanged dieldrin (2.581 ppm) and photodieldrin (0.029 ppm).

6.3.2.4 Other Media

No studies were located regarding the degradation or transformation of aldrin or dieldrin in other media.

6.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

Aldrin is readily converted to dieldrin in the environment. Dieldrin is subject to atmospheric transport, and, as a result, is ubiquitous in the environment. Dieldrin persists because it is relatively resistant to biotransformation and abiotic degradation. Thus, it is found in low levels in all media (air, water, and soil).

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6.4.1 Air

Aldrin and dieldrin enter the atmosphere through various mechanisms such as spray drift during application of the compounds as insecticides, water evaporation, and suspension of particulates to which the compounds are absorbed. The analysis of 2,479 air samples from 16 states from 1970 to 1972 revealed the following ambient concentrations: aldrin, mean 0.4 ng/m^3 (3×10^{-5} ppb), 13.5% of samples positive; dieldrin, mean 1.6 ng/m^3 (1×10^{-4} ppb), 94% of samples positive (Kutz et al. 1976).

The annual atmospheric deposition of dieldrin to the five Great Lakes was estimated based on measurements taken in the late 1970s (Eisenreich et al. 1981). The results indicated that 0.54, 0.38, 0.55, 0.17, and 0.13 metric tons/year were deposited into Lake Superior, Lake Michigan, Lake Huron, Lake Erie, and Lake Ontario, respectively. The annual mean gas-phase, particulate-phase, and precipitation concentrations of dieldrin were studied over the U.S. Great Lakes from 1990 to 1992 (Hoff et al. 1996). The annual mean gas-phase concentrations of dieldrin over Lakes Superior, Michigan, Erie, and Ontario were 14, 34, 30, and 23 pg/m^3 , respectively. The particulate-phase concentrations of dieldrin over Lakes Superior, Michigan, Erie, and Ontario were 1.5, 1.9, 3.2, and 1.6 pg/m^3 , respectively. Finally, the concentrations of dieldrin in precipitation falling over Lakes Superior, Michigan, Erie, and Ontario were 0.4, 0.99, 0.8, and 0.6 ng/L , respectively. The total wet deposition of dieldrin in 1992 for Lakes Superior, Michigan, Erie, and Ontario was 21, 58, 28, and 11 kg, respectively. More recent data on the atmospheric concentrations of aldrin and dieldrin were gathered in 1986, approximately 10 years after the use of aldrin and dieldrin was restricted in the Great Lakes Basin (Chan and Perkins 1989). It was found that aldrin was present in 5 of 75 wet precipitation samples at three of four sampling sites located around the basin. Two of the three sites had a mean concentration of 0.01 ng/L (1.0×10^{-5} ppb), while the third site had a mean concentration of 0.24 ng/L (2.4×10^{-4} ppb). Dieldrin was detected at all four sites and in >60% of the samples at mean concentrations ranging from 0.41 to 1.81 ng/L (4.1×10^{-4} – 1.8×10^{-3} ppb). The highest concentrations of both aldrin and dieldrin were found in samples collected at Pelee Island at the western end of Lake Erie (maximum concentrations of 3.4 ng/l [3.4×10^{-3} ppb] and 5.9 ng/L [5.9×10^{-3} ppb], respectively). In 1979–1980, dieldrin was detected in the ambient air and rainfall over College Station, Texas, at average concentrations of 0.08 ng/m^3 (5.1×10^{-6} ppb) and 0.80 ng/L (8×10^{-4} ppb), respectively (Atlas and Giam 1988). The washout ratio (concentration in rain/concentration in air) for dieldrin was calculated to be 8.9. Dieldrin was present in rainfall measured at three points in Canada during 1984, at mean concentrations of 0.78 ng/L (7.8×10^{-4} ppb) over Lake Superior, 0.27 ng/L in New Brunswick, and 0.38 ng/L (3.8×10^{-4} ppb) over northern Saskatchewan (Strachan 1988).

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Between 1991 and 1993, 18 fogwater samples, 31 rainwater samples, and 17 atmosphere (gas and particles) samples were analyzed for aldrin and dieldrin from a rural area in Colmar, east of France (Millet et al. 1997). The mean concentrations of aldrin and dieldrin in fogwater collected from 1991 to 1993 were 3.5 and 5 ng/mL, respectively. The mean concentrations of particle bound aldrin and dieldrin in fogwater collected from 1991 to 1993 were 15 and 17 ng/mL, respectively. The mean concentrations of aldrin and dieldrin in rainwater collected in 1992 were 0.05 and 0.5 ng/ml, respectively. The mean concentration for both aldrin and dieldrin in the vapor-phase collected in 1992 was the same at 0.7 ng/cm³. Finally, the mean concentrations of aldrin and dieldrin in the particulate phase collected in 1992 were 0.6 and 0.7 ng/mL, respectively.

The atmospheric concentration of both aldrin and dieldrin were studied in the National Park of Ordesa, Spain from April to August 1995 (Nerin et al. 1996). The study found that on April 10 and August 23, the concentration of aldrin was below detection limit (1 pg/m³) while on June 23, the concentration of aldrin was 12 pg/m³. The concentration of dieldrin on April 10 was also below the detection limit (1 pg/m³), but was detected at a concentrations of 6 and 3 pg/m³ on June 23 and August 23, respectively.

6.4.2 Water

A comprehensive study of U.S. drinking water samples (1975) revealed that <17% of the samples contained dieldrin, with 78% of the positive samples containing concentrations between 4 and 10 ng of dieldrin per L of water (0.004–0.01 ppb) (EPA 1980a). In a recent study, the concentration of various pesticides were measured six times from September 1995 to September 1996 in drinking water samples from 80 randomly selected residences of Maryland (MacIntosh et al. 1999). Dieldrin was not detected in any of the samples taken during this test (limit of detection=25 µg/L). Between November 1, 1983 and July 1, 1992, the California EPA tested various wells for pesticide residues throughout the state of California (California EPA 1995). Aldrin and dieldrin were not detected in any of the 1,304 wells (covering 33 counties) sampled during this study. In another study, dieldrin residues were analyzed for in 208 well water samples collected from nine urban areas from across the United States (Kolpin et al. 1997). Dieldrin was detected in 2.4% of wells samples (detection limit=0.005 µg/L) at a maximum concentration of 0.045 µg/L. Along the north coast of Australia, 659 water samples were surveyed for pesticide residues (McDougall et al. 1994); 20% of storage tanks of domestic water supplies were contaminated with dieldrin at or above 0.05 µg/L.

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In earlier studies, dieldrin was found more often than any other pesticide in water samples collected from all major river basins (mean concentration, 7.5 ng/L [0.0075 ppb]) in the United States (Weaver et al. 1965). In 1976, dieldrin was reported in many fresh surface waters of the United States with mean concentrations ranging from 5 to 395 ng/L (0.005–0.395 ppb) (EPA 1980a). Data maintained in the STORET database for 1980–1982 included aldrin and dieldrin concentrations in industrial effluent, ambient water, sediments, and biota. Median values from all STORET stations were as follows (Staples et al. 1985):

Media	Aldrin		Dieldrin			
	Median (ppb)	Number of samples	Percentage detectable	Median (ppb)	Number of samples	Percentage detectable
Effluent	<0.01	677	3.1	<0.01	676	3.7
Water	0.001	7,891	40.0	0.001	7,609	40.0
Sediment	0.1	2,048	33.0	0.8	1,812	33.0
Biota	<0.1	211	0	0.03	530	41

Influent and effluent samples from New York City's 14 water pollution control plants were collected and analyzed six times during the course of 5 years (1989–1993) to determine the concentration of chemical contaminants (Stubin et al. 1996). Of the 168 samples collected, aldrin was detected in 12 influent water samples in 1990, once in 1992 and 9 times in 1993. The concentration of aldrin in influent samples ranged from 0.024 to 1.1 µg/L. Aldrin was also detected in effluent samples 11 times in 1990, twice in 1992, and six times in 1993. The concentration of aldrin in effluent samples ranged from 0.008 to 0.44 µg/L. Dieldrin, however, was not detected in any influent samples, and was only detected in two effluent samples taken in 1993. The concentration of dieldrin in the effluent samples ranged from 0.012 to 0.028 µg/L.

In 1980, aldrin and dieldrin were detected in water samples taken from the Inner Harbor Navigation Canal of Lake Pontchartrain (New Orleans, Louisiana) on the ebb and flood tides at a depth of 1.5 meters; respective concentrations were 0.3 ng/L (0.0003 ppb) and 5.6 ng/L (0.0056 ppb) for aldrin and 0.6 ng/L (0.0006 ppb) and 5.9 ng/L (0.0059 ppb) for dieldrin (McFall et al. 1985). In 1987, dieldrin was detected in seawater samples taken from the Gulf of Mexico at concentrations ranging from 0.009 to 0.02 ng/L (9×10^{-6} – 2×10^{-5} ppm) and from seawater off the southeastern United States at 0.007–0.01 ng/L (7×10^{-6} – 1×10^{-5} ppm); aldrin was also detected in the southeastern U.S. coastal waters at concentrations of 0.31–1.5 ng/L (0.0003–0.001 ppb) (Sauer et al. 1989). Aldrin and dieldrin were detected in water and

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sediment samples taken between 1975 and 1980 at 160–180 stations on major rivers of the United States as part of the National Pesticide Monitoring Program. Aldrin and dieldrin were both detected in 0.2% of the 2,946 water samples and in 0.6 and 12% of the approximately 1,016 sediment samples, respectively (USGS 1985). In 1988, dieldrin was detected in 9% of 422 groundwater samples taken from a sandy, alluvial aquifer in Illinois at a median concentration of 0.01 $\mu\text{g/L}$ (1.0×10^{-5} ppb), and in 4% of groundwater well samples taken in the vicinity of an agricultural dealer facility, at a mean concentration of 0.03 $\mu\text{g/L}$ (3.0×10^{-5} ppb) (Hallberg 1989). Out of 2,459 sites from the largest river basins and aquifers in the United States tested between 1992 and 1996, dieldrin had a frequency of detection of 1.63% and a maximum concentration of 0.068 $\mu\text{g/L}$ (Koplin et al. 2000).

Analysis of urban storm water runoff collected between 1979 and 1983 in the Canadian Great Lakes Basin found dieldrin to be present in approximately 32 of 124 water samples at a mean concentration of 5.1×10^{-4} $\mu\text{g/L}$ (5.1×10^{-4} ppb) and in approximately 17 of 110 runoff sediment samples at a mean concentration of 4.4×10^{-3} mg/kg (4.4 ppb). Aldrin was found in approximately 13 of 129 runoff sediment samples at a mean concentration of 1.2×10^{-3} mg/kg (1.2 ppb) but was not detected in any water samples (Marsalek and Schroeter 1988). These concentrations resulted in mean annual loadings to the Canadian Great Lakes Basin of 0.2 kg/year for aldrin and 0.6 kg/year for dieldrin. In 1982, water samples taken from 19 U.S. cities for the National Urban Runoff Program, found aldrin to be present only in samples taken from Washington, D.C., at a concentration of 0.1 $\mu\text{g/L}$ (0.1 ppb) (6% of samples), and dieldrin was detected only in water from Bellevue, Washington, at 0.008–0.1 $\mu\text{g/L}$ (0.008–0.1 ppb) (2% of samples) (Cole et al. 1984). Water sampling conducted during the 1986 spring isothermal period in the Great Lakes did not detect aldrin in any samples. Dieldrin, however, was present in all samples at mean concentrations ranging from 0.300 ng/L (0.0003 ppb) for Lake Superior to 0.402 ng/L (4.2×10^{-4} ppb) in Lake Erie (Stevens and Neilson 1989).

Aldrin was identified in leachate from the Love Canal industrial landfill in Niagara Falls, New York, at a concentration of 0.023 mg/L (23 ppb) (data were gathered prior to 1982) (Brown and Donnelly 1988). In 1986, a waste site was identified in Clark County, Washington, that contained buried drums believed to have originally held chemicals used at a plywood manufacturing plant. Analysis of the soil and water contamination found aldrin to be present in groundwater samples taken from shallow wells on site at a maximum concentration of 2.12 $\mu\text{g/L}$ (2.12 ppb) and in groundwater samples from nearby private wells at 0.79 $\mu\text{g/L}$ (0.79 ppb) (EPA 1986i). At a hazardous waste site in Gallaway, Tennessee, drums and bottles containing chemicals from a pesticide blending operation had been emptied or discarded into a number of

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small ponds on the site. Dieldrin was present in on-site surface waters at 0.40–1.4 ppb, but was not detected in any off-site water samples (EPA 1987i).

6.4.3 Sediment and Soil

As a result of the rapid conversion of aldrin to dieldrin, soil residues of dieldrin are found in higher concentration and with greater frequency than residues of aldrin, even though aldrin was applied more frequently to the soil. The amount of dieldrin and aldrin residues in soils was monitored from 12 separate farm lands located in the Fraser Valley of British Columbia, Canada in 1989 (Szeto and Price 1991). Each farm had a known history of at least 25 years of vegetable growing and use of various pesticides. Aldrin was detected on one farmland with muck soil at a mean concentration of 78 ppb dry weight, while dieldrin was detected on two farmlands containing muck soils at a mean concentration of 692 ppb dry weight (range from 104 to 1,280). In a separate study, the concentration ranges of dieldrin in agricultural soil samples taken in 1995 and 1996 from Alabama, Ohio, Indiana, and Illinois were not detected (nd)–23, nd–4250, nd–69, and nd–13 ng/g dry weight, respectively (Bidleman 1999).

An analysis of sediment samples taken from Lake Ontario in 1981 showed that dieldrin levels had increased from approximately 26 ng/g (26 ppb) in 1970 to 48 ng/g (48 ppb) in 1980, although the use of dieldrin was banned in much of the Great Lakes Basin in the early 1970s (Eisenreich et al. 1989). The National Soils Monitoring Program (Kutz et al. 1976) detected dieldrin in soils at varying concentrations and areas throughout 24 states; the mean concentration ranged from 1 to 49 ppb. At a hazardous waste site in Gallaway, Tennessee, drums and bottles containing chemicals from a pesticide blending operation had been emptied or discarded into a number of small ponds on the site. Dieldrin was present in sediment samples from on-site ponds at 1,400 ppb and in one off-site sediment sample (concentration not specified) (EPA 1987i). Sediment samples taken from two lakes near the U.S. Army Rocky Mountain Arsenal, Colorado in 1983, indicated that aldrin and dieldrin persisted in the sediments long after deposition ceased. Concentrations up to 2,050 ppb for aldrin and 100 ppb for dieldrin at a core depth of approximately 21 cm were found in one lake. A second lake also had elevated levels of aldrin and dieldrin contamination, but at lower concentrations (approximately 250 ppb for aldrin and 40 ppb for dieldrin) and at a lower core depth, indicating that most of the deposition had occurred at an earlier date (Bergersen 1987). The concentration of dieldrin and aldrin was also studied in sediment samples from three coastal lagoons in the southeast of the Gulf of Mexico (Botello et al. 1994). The average concentrations of aldrin in sediment samples taken from the Carmen, Machona, and Alvarado lagoons were 0.70, 1.15, and 2.11 ng/g dry weight, respectively. The average concentrations of dieldrin in

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sediment samples taken from the Carmen, Machona, and Alvarado lagoons were 6.84, 0.59, and 2.05 ng/g dry weight, respectively. A monitoring survey of 17 wetland areas in the north central United States, found dieldrin to be present in only one Iowa sediment sample at 170 ng/g (170 ppb) dry weight (Martin and Hartman 1985).

6.4.4 Other Environmental Media

The persistence of dieldrin in the environment is demonstrated by a monitoring survey conducted in and around cotton fields in four counties in Alabama between 1972 and 1974. Although cotton farmers had not used aldrin or dieldrin "for several years," dieldrin was found to be present at 7–40 ppb in 50% of the soil samples; at <100 ppb in 50% of forage samples with levels declining over time; at an average concentration of 1,490 ppb in 11 of 19 rat tissue samples with number of positive samples increasing between 1973 and 1974; at low levels in some quail tissue samples (maximum level=790 ppb); at levels declining from 302 to 70 ppb between 1972 and 1974 for mockingbird tissue samples; and at <30 ppb in most of the 25% positive fish tissue samples taken from farm ponds (Elliott 1975). Aldrin was estimated to have a half-life of 1.7 days on crops with the half-life of dieldrin ranging from 2.7 to 6.8 days depending on the crop and formulation (Willis and McDowell 1987). These half-life values were based on the disappearance of aldrin and dieldrin due to volatilization, adsorption to plant surfaces, relative humidity, rain, wind, temperature, and sunlight.

Dieldrin was detected in the liver and fat of arctic ground squirrels trapped near three lakes located at the foothills of the Brooks Range, Alaska between 1991 and 1993 (Allen-Gil et al. 1997). The mean concentrations of dieldrin in squirrel liver from Elusive Lake (seven samples), Feniak Lake (seven samples), and Schrader Lake (seven samples) were 10.91, 1.53, and 14.42 $\mu\text{g/g}$ wet weight, respectively. The mean concentrations of dieldrin in squirrel fat from Elusive Lake (no samples), Feniak Lake (seven samples), and Schrader Lake (five samples) were below the minimum detectable limit, 0.0, and 0.5 $\mu\text{g/g}$ wet weight, respectively.

Blood samples were collected and analyzed for dieldrin concentrations from nestling bald eagles at active nests in the Canadian portion of the Great Lakes Basin from 1990 to 1994 (Donaldson et al. 1999). Mean dieldrin concentrations in eagle blood samples taken from Lake Erie, Huron, Nipigon, Superior, and Woods were 0.003 (30 samples), 0.007 (1 sample), 0.0031 (7 samples), 0.0051 (11 samples), and 0.0031 mg/kg wet weight (2 samples), respectively. Residue levels of dieldrin in unhatched bald eagle eggs collected along Lake Erie from 1974 to 1980 (six samples) and from 1989 to 1994 (six samples)

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were 1.28 and 0.49 mg/kg wet weight, respectively. Seven bald eagle eggs from the Tanana River, Alaska collected in 1990 and 1991 contained dieldrin with a mean concentration of 0.028 ppm (Ritchie and Ambrose 1996). Peregrin falcon eggs from Rankin Inlet collected from 1991 to 1994 (20 samples) and from 1982 and 1986 (36 samples) contained dieldrin at mean concentrations of 0.361 (range of 0.13–1.66) and 0.41 (range of 0.045–1.80) $\mu\text{g/g}$ wet weight, respectively (Braune et al. 1999). Osprey eggs collected at five locations on the Fraser River from 1991 to 1997 contained dieldrin; the highest concentrations were reported at the Fraser River site below Quesnel with a mean value of 5.2 $\mu\text{g/kg}$ wet weight (Elliott et al. 2000). Lower concentrations were reported for eggs collected at the other locations with mean values generally <2 $\mu\text{g/kg}$ wet weight. dieldrin was detected with a mean concentration of 0.25 $\mu\text{g/g}$ in 75 out of 312 double-crested cormorant eggs and embryos collected from Cat Island, Green Bay, Wisconsin in 1994 and 1995 (Custer et al. 1999). The mean concentration of dieldrin in tree swallow eggs and tree swallow nestlings collected in 1998 at Pigeon Creek, Iowa (three samples); Duck Creek, Iowa (three samples); and Lindsey Harbor, Iowa (seven samples) along the Upper Mississippi River was 0.03 $\mu\text{g/g}$ wet weight (Custer et al. 2000).

Waterfowl from Northern Canada were collected from 1988 to 1995 and divided into browsers, grazers, omnivores, molluscivores, and piscivores (Braune et al. 1999). The highest concentrations of dieldrin were found in tissues of waterfowl feeding at the upper trophic levels. Concentrations of dieldrin ranged from nd to 5.0 ng/g wet weight, nd to 3.2 ng/g wet weight, nd to 15.9 ng/g wet weight, nd to 120 ng/g wet weight, and nd to 54.7 ng/g wet weight, respectively.

Mean concentrations of dieldrin in snapping turtle eggs collected at four sites along the St. Lawrence River in the Mohawk territory of Akwesasne during June, 1998 ranged from 4 to 280 ng/g wet weight with an overall mean concentration of 38.13 ng/g wet weight (de Solla et al. 2001). Aldrin and dieldrin were detected in the plasma of juvenile alligators from three lakes in central Florida (Guillette et al. 1999). The mean concentrations of dieldrin in males from Lake Woodruff, Lake Apopka, and Orange Lake were 0.24, 1.68, and 0.75 ng/mL plasma, respectively. The mean concentrations of dieldrin in females from these lakes were 0.31, 2.87, and 0.39 ng/mL plasma, respectively. The mean concentration of aldrin for juvenile alligators in all three lakes was 0.34 ng/mL plasma.

In 1985, fish samples taken from the lower Savannah River in Georgia and South Carolina were found to occasionally contain dieldrin but at concentrations of <0.01 $\mu\text{g/g}$ (10 ppb) (Winger et al. 1990); common carp and white bass samples from a lake in Kansas located in an agricultural area had mean concentrations of 0.069 and 0.058 ppb, respectively (Arruda et al. 1988). Fish samples taken from

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tributary rivers around the Great Lakes in 1980–1981 had dieldrin levels up to 0.15 mg/kg (150 ppb) (average concentration=0.03 mg/kg [30 ppb]) (DeVault 1985). Fish taken from Lake Huron between 1970 and 1980 had mean dieldrin concentrations ranging from 0.01 to 0.50 mg/kg (10–500 ppb) (EPA 1985e); however, by 1984, mean concentrations of dieldrin in Lake Michigan coho salmon had dropped to 0.01 µg/g (10 ppb) from 0.06 µg/g (60 ppb) in 1980 (DeVault et al. 1988). An analysis of 315 composite samples of whole fish collected from 107 sites nationwide in 1980–1981 as part of the National Pesticide Monitoring Program found that the mean concentrations of dieldrin were essentially unchanged since 1978–1979. In 1978, dieldrin was detected in 81% of the samples, and in 1980, in 75% of the samples at mean concentrations of 0.05 µg/g (50 ppb) wet weight and 0.04 µg/g (40 ppb), respectively (Schmitt et al. 1985). Three of eight samples of bluegill (*Lepomis macrochirus*) collected from the San Joaquin Valley in July 1981 contained dieldrin at concentrations ranging from 0.005 to 0.008 mg/kg (5–8 ppb) wet weight; four of the eight common carp (*Cyprinus carpio*) obtained from the same sites contained dieldrin at concentrations ranging from 0.015 to 0.067 mg/kg (15–67 ppb) wet weight (Saiki and Schmitt 1986). Dieldrin was also detected in a variety of fish taken from a section of Lake Oconee in Georgia that received storm runoff from insecticide-treated areas between 1981 and 1982. Dieldrin concentrations ranged from <10 to 200 µg/kg (10–200 ppb). Dieldrin was not detected in fish taken from the lake after 1982 (Bush et al. 1986). A survey of 17 wetland areas in the north central United States found dieldrin in two fish samples taken from Kansas and Iowa at concentrations of 6 ng/g (6 ppb) and 9 ng/g (9 ppb), respectively (Martin and Hartman 1985). Dieldrin was found in 5 of 20 raw bluefish fillets collected in Massachusetts waters in 1986, at concentrations of 0.02–0.04 ppm (20–40 ppb); after cooking, dieldrin was still detected in the fillets, indicating that heating does not degrade the pesticide in foods (Trotter et al. 1989). Aldrin and dieldrin were detected in shrimp (*Penaeus setiferus* and *Penaeus aztecus*) collected from the Calcasieu River Basin in an industrial area of Louisiana in 1985–1986. Aldrin was present in shrimp taken from 7 of 30 stations at concentrations ranging from 0.01 to 0.12 µg/g (10–120 ppb), and dieldrin was present in 21 of 30 samples at concentrations of 0.05–9.47 µg/g (50–9,470 ppb) (average concentration 1.57 µg/g [1,570 ppb]) (Murray and Beck 1990). Between October 1981 and September 1986, over 12,044 imported and 6,391 domestic commodities were sampled for pesticide residues. Dieldrin was detected in 420 imported and 44 domestic products; however, the tolerance (the maximum amount of a residue expected in a food when a pesticide is used according to label directions, provided that the level does not present an unacceptable health risk) for dieldrin was exceeded in only eight imported products and one domestic product, indicating that most agricultural products do not contain harmful levels of dieldrin (Hundley et al. 1988).

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The concentrations of dieldrin and aldrin were studied in bivalve mollusks obtained from three coastal lagoons in the southeast of the Gulf of Mexico (Botello et al. 1994). The average concentration of aldrin in bivalve mollusks collected from the Carmen, Machona, and Alvarado lagoons were 2.56, 1.61, and 6.61 ng/g dry weight, respectively. Dieldrin was not detected in any bivalve mollusks collected from the Carmen, Machona, and Alvarado lagoons. Dieldrin concentrations were analyzed for in nine marine mammal species samples collected in 1987 (Becker et al. 1997). The means and standard deviations of dieldrin in northern fur seal, ringed seal, pilot whale, harbor porpoise, beluga whale from the Arctic, and beluga whale from Cook Inlet were 13.6, 43.2 ± 53.8 , 262 ± 240 , 963 ± 294 , 290 ± 106 , and 105 ± 66.2 ng/g wet weight, respectively.

One study examined persistent organochlorines concentrations in blubber samples from 16 dead beluga whales collected during 1993–1994 in the St. Lawrence River estuary (Muir et al. 1996). The mean concentrations of dieldrin in seven female and nine male beluga whales were 1,360 ng/g lipid weight (lw) (range=326–2,360 ng/g lw) and 2,020 ng/g lw (range=1,440–2,620 ng/g lw), respectively. The study found a temporal upward trend in dieldrin concentration in female beluga whales and slightly augmented levels of dieldrin in males. The average dieldrin concentration in female beluga whales in 1987 was 450 ng/g lw while in males, the average concentration of dieldrin measured from 1986 to 1988 was 1,650 ng/g lw. In a separate study, biopsies were collected from Right whales in the Bay of Fundy in 1994 (30 samples), 1995 (17 samples), and 1996 (15 samples) and at sites in Georgia and Cape Cod Bay in 1997 (Weisbrod et al. 2000). For each collection period mean concentrations were 513 and 93 ng/g sample lipid content and nd, respectively, for aldrin and 1,141, 1,349, and 4,244 ng/g sample lipid content, respectively, for dieldrin. Zooplankton samples collected in 1995 and 1996 from Georges Bank, Bay of Fundy, and Cape Cod Bay contained aldrin at concentrations that were undetectable to 8.9 ng/g sample lipid content and dieldrin at concentrations that were undetectable to 23 ng/g sample lipid content.

Dieldrin concentrations were determined in archived samples of whole lake trout collected yearly from eastern Lake Ontario between 1977 and 1993 (Huestis et al. 1996). The mean concentrations of dieldrin in trout collected in 1977, 1980, 1983, 1986, 1989, 1990, 1992, and 1993 were 313, 218, 135, 103, 97.3, 99.0, 73.1, and 78.4 ng/g, respectively. An investigation of the temporal trends of pesticide residues in fish from Lake Michigan indicated a decrease in dieldrin concentrations from 1982 to 1990 (Miller et al. 1992). Total dieldrin concentrations in lake trout decreased 68% from 410 ± 50 $\mu\text{g}/\text{kg}$ in 1982 to 130 ± 30 $\mu\text{g}/\text{kg}$ in 1990. In Lake Superior, dieldrin concentrations in fish did not appear to as much over a 3-year period. The total dieldrin concentration in lake trout in 1982 was 50 ± 10 $\mu\text{g}/\text{kg}$ while in 1985, the concentration was 40 ± 10 $\mu\text{g}/\text{kg}$. In a separate study (Zabik et al. 1996), concentrations of dieldrin in

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skin-off lake trout collected from both Lake Huron and Lake Michigan, and siscowets from Lake Superior were reported as 0.029, 0.076, and 0.027 ppm, respectively.

During the fiscal years 1989 to 1994, the U.S. Food and Drug Administration (FDA) collected and analyzed 545 domestic surveillance samples of mixed feed rations for pesticide residues (Lovell et al. 1996). The mixed feed rations represented feed fed to cattle, poultry, swine, pets, fish, and other miscellaneous animals. The results indicated that dieldrin was detected in five samples (three trace, two quantifiable) at 10 µg/kg.

Lichens collected in the Arctic between 1993 and 1994 contained detectable residues of dieldrin; concentrations of below detection to 0.72 ng/g dry weight were reported with the highest concentrations found in samples collected from Makinson Inlet and King Edward Point in the Northwest Territories (Braune et al. 1999). Saxifrage samples from Ellesmere Island and Axel Heiberg Island, collected in 1990, contained dieldrin at mean concentrations of 0.46 and 0.44 ng/g dry weight, respectively.

6.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

Use of aldrin and dieldrin for pest control on crops such as cotton, corn, and citrus products was canceled by the EPA in 1974 (EPA 1974a), while use for extermination of termites was voluntarily canceled by the manufacturer in 1987 (EPA 1990b). However, during the period of widespread use and production of aldrin and dieldrin, intake by workers who manufactured these compounds was estimated to range from 0.72 to 1.10 mg/person/day with a good correlation between levels in tissue (fat, serum, and urine) and total length of exposure or intensity of exposure (Hayes and Curley 1968). The National Occupational Exposure Survey, conducted by NIOSH between 1980 and 1983, estimated that 647 employees were exposed to aldrin and 760 employees were exposed to dieldrin in the workplace (NOES 1990). One pest control operator was found to have 0.5 and 0.3 µg dieldrin on his left and right hands, respectively, >2 years after his last exposure to aldrin; serum blood levels taken at the same time showed 10 ppb dieldrin. A further analysis of individuals exposed to dieldrin found no correlation between the pesticide levels on their hands and in their sera (Kazen et al. 1974). A 1981 survey of Florida citrus field workers found dieldrin to be present in >3% of the 567 serum samples, at a mean concentration of 1.8 ppm (1,800 ppb) (Griffith and Duncan 1985). Workers cleaning up hazardous waste sites may also be exposed, but no information on monitored levels of exposure was found.

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In one pilot study, food, beverage, and biological specimens (blood and urine) were collected and analyzed for pesticides from six farm families living in Iowa and North Carolina (Brock et al. 1998). Although dieldrin concentrations were below detection limits (0.23 ng/mL) in five of the families studied, one family in particular had elevated levels. One farmer from Iowa had a mean dieldrin concentration of 20.55 ± 2.61 ng/mL while that person's spouse had a mean concentration of 7.52 ± 0.68 ng/mL. Solid food samples from this farm also contained elevated levels of dieldrin ranging from 15.0 to 28.0 ng/g. On the other five farms, dieldrin concentrations in solid food samples were below the detection limit (0.75 ng/g). Finally, dieldrin levels in beverages were below the detection limit in five of the farm families studied except for the one family from Iowa with elevated dieldrin levels, which had an average concentration of 11.0 ng/g.

The National Health and Nutrition Examination Survey (NHANES II) conducted between 1976 and 1980, found that an estimated 10.6% of the population aged 12–74-years-old, were exposed to dieldrin based on an analysis of blood serum and urine specimens (Stehr-Green 1989). When specimens from populations in the northeast, midwest and south regions of the United States were examined almost 20% of the adults aged 45–74 years had quantifiable levels of dieldrin (mean concentration 1.4 ppb), while only 1.5% of the adults aged 12–24-years-old had quantifiable levels (mean concentration 1.4 ppb) (Murphy and Harvey 1985; Stehr-Green 1989). Dieldrin was found in 14 of 46 adipose tissue samples taken from cadavers and surgical patients during the 1982 Human Adipose Tissue Survey conducted by EPA on a nationwide basis. Concentrations of dieldrin in wet tissue were in trace amounts ranging from 0.053 to 3.84 $\mu\text{g/g}$ (53–3,840 ppb) (mean concentration, 0.458 $\mu\text{g/g}$ or 458 ppb). Aldrin was not present in any of the samples; the detection level was 0.010 $\mu\text{g/g}$ (10 ppb) for a 20-g tissue sample (EPA 1986a). In 1976 and 1984, human adipose tissue samples were taken from cadavers of Canadians from the Great Lakes region and examined for the presence of a variety of compounds. Dieldrin was found in 100% of the tissue samples taken each year at a mean concentration of 0.049 $\mu\text{g/g}$ (49 ppb) wet weight in 1976 (Mes et al. 1982) and 0.047 $\mu\text{g/g}$ (47 ppb) wet weight in 1984 (Williams et al. 1988). Adipose tissue collected from 46 infertile women in Belgium between 1996 and 1998 contained dieldrin at a mean concentration of 13.1 ± 6.6 ng/g. Dieldrin was not detected in the serum of the women (Pauwels et al. 2000). Based on a study with 12 male volunteers who ingested up to 225 μg dieldrin per day for up to 2 years, a wet weight BCF of 30 was calculated, although the BCF for the lipid fraction of body weight was 45. Other studies have found wet weight BCFs ranging from 38 to 77 (mean, 48.7) and lipid basis BCFs ranging from 55 to 115 (mean, 70.9) (Geyer et al. 1986, 1987). Blood samples taken from residents of El Paso, Texas, during 1982–1983, showed aldrin to be present in 39 of 112 samples (34%) at a mean concentration of 4.6 ppb (Mossing et al. 1985).

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Individuals living in homes contaminated by past termiticide treatment constitute a significant group exposed to aldrin and dieldrin in indoor air. Measurements of air concentrations in homes 1–10 years after termiticide treatment showed dieldrin levels ranging from 0.0006 to 0.03 ppb in living rooms and bedrooms and all interior areas (Dobbs and Williams 1983). Air samples were taken and analyzed for aldrin over the course of 6 months from 29 dwellings treated with aldrin for prevention of termite infestation (Gun et al. 1992). Blood samples were also analyzed for dieldrin levels of one occupant from each dwelling. The concentration of atmospheric aldrin was recorded for the first six months of the study. Prior to treatment, the median concentration of aldrin was $0.044 \mu\text{g}/\text{m}^3$; 1 week post-treatment $2.6 \mu\text{g}/\text{m}^3$; 6 weeks post-treatment, $0.72 \mu\text{g}/\text{m}^3$; and 6 months post-treatment, $0.57 \mu\text{g}/\text{m}^3$. Prior to treatment, the median concentration of dieldrin in blood was $0.75 \text{ ng}/\text{mL}$ while 3 months post-treatment the median concentration was $1.2 \text{ ng}/\text{mL}$.

The levels of aldrin and dieldrin were monitored in human blood samples taken from the general population from the rural town of Ahmedabad, India (Bhatnagar et al. 1992). Blood samples from 31 male subjects, ages 18–57 (mean 28.4 years), were collected from 1989 to 1990. The concentration of aldrin and dieldrin ranged from 0 to $0.813 \mu\text{g}/\text{L}$ (mean $0.200 \mu\text{g}/\text{L}$) and from 0 to $3.730 \mu\text{g}/\text{L}$ (mean $2.152 \mu\text{g}/\text{L}$), respectively.

A pilot study of non-occupational general population exposure to pesticides in ambient air inside and outside the home was conducted in nine homes in Florida in August 1985. Air was monitored for 24 hours outside the house and inside the house, and personal air monitors were worn by one occupant of each house. Aldrin and dieldrin were detected in indoor air at 6 and 5 of the 9 households, respectively; outdoors at 4 of the 9 households each; and by personal monitors for 3 and 5 of the 9 individuals, respectively. In one designated high-pesticide-use household, aldrin and dieldrin were detected in the indoor air at average concentrations of $0.058 \mu\text{g}/\text{m}^3$ (0.004 ppb) and $0.038 \mu\text{g}/\text{m}^3$ (0.002 ppb), respectively. Neither compound was detected in the outdoor air immediately adjacent to the home, and concentrations detected with personal air monitors were half (aldrin) to one-third (dieldrin) the concentrations for ambient indoor air (Lewis et al. 1988). A composite sample of the dust from four Seattle homes collected in 1988–1989 showed dieldrin to be present at 1.1 ppm, although none of the homeowners could remember using the pesticide. It was suggested that the source of the dieldrin was soil surrounding the homes; however, since the use of dieldrin is restricted to termite control, and Seattle has few termites, the source of the contaminated soil is unknown (Roberts and Camann 1989).

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Atmospheric sampling of aldrin and dieldrin conducted from 1970 to 1972 indicated that aldrin and dieldrin were present at mean concentrations of 0.4 ng/m^3 (2.7×10^{-5} ppb) and 1.6 ng/m^3 (1.02×10^{-4} ppb), respectively (Kutz et al. 1976). Combining these figures and assuming that 20 m^3 of air are inspired each day, average daily intake of aldrin plus dieldrin from the atmosphere would be 0.57 ng/kg body weight in 1972. However, the cancellation of the use of these compounds suggests that current inhalation intake will be much less. Guicherit and Schulting (1985) used data on air samples collected in the western part of the Netherlands in 1979–1981 and calculated the average daily intake by inhalation to be 0.02 ng dieldrin/kg body weight and 0.01 ng aldrin/kg body weight.

A significant source of general population exposure to dieldrin is through diet. In the absence of occupational or domestic use as a pesticide, food is probably the primary source of dieldrin residues in human adipose tissues (Ackerman 1980). Because of the rapid epoxidation of aldrin in the environment, it is not considered to be an important human dietary contaminant, with an average intake of $<0.001 \text{ } \mu\text{g/kg/day}$. Dieldrin, however, may be ingested as a result of eating contaminated fish, milk, and other foods with a high fat content including meat. EPA established tolerances for aldrin and dieldrin in or on raw agricultural commodities at maximums of $0.0\text{--}0.1 \text{ ppm}$, depending on the crop (Sittig 1980). Table 6-1 shows a summary of dieldrin residues in adult dietary components analyzed in 1981–1982 (Gartrell et al. 1986a). A 1985 Canadian survey of foods found that although aldrin was not detected in any of the food samples analyzed, dieldrin was detected in all food composites at $0.00011 \text{ } \mu\text{g/g}$ in fruit; $0.0019 \text{ } \mu\text{g/g}$ in milk; $0.0031 \text{ } \mu\text{g/g}$ in leafy vegetables, eggs, and meat; and $0.023 \text{ } \mu\text{g/g}$ in root vegetables (Davies 1988). Dieldrin residues may persist in foods such as milk butterfat and subcutaneous fat in cattle with an estimated half-life in butterfat of 9 weeks (Dingle et al. 1989). Samples of ultra-pasteurized heavy cream and cow's milk purchased in Binghamton, New York, in 1986 had dieldrin levels of 0.006 and 0.003 ppm , respectively (Schechter et al. 1989a).

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Table 6-1. Dieldrin Residues in Adult Dietary Components (1980–1982)^a

Food group	Residue range (ppm)	Average concentration (µg/day)
Dairy	Trace to 0.003	0.0006
Fish, poultry, meat	Trace to 0.004	0.0012
Potatoes	Trace to 0.002	0.0004
Root vegetables	Trace to 0.005	0.0004
Leaf vegetables	Trace to 0.002	0.0002
Legumes	ND	ND
Garden fruits	Trace to 0.011	0.0021
Fruits	0.001	0.0001
Cereals and grain	0.004	0.0001
Oils and fats	Trace to 0.002	0.0003
Sugar	ND	ND
Beverages	ND	ND

^aDerived from Gartrell et al. 1986a

ND = not detected

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During the period of 1965–1970, total U.S. dietary intake was reported to be 0.05–0.08 ug dieldrin/kg/day and 0.0001–0.04 ug aldrin/kg/day (IARC 1974b). Since 1970, the use of aldrin and dieldrin on food has been cancelled, and dietary intake has decreased. An FDA Total Diet Study, conducted between 1982 and 1984, found that aldrin intake was <0.001 µg/kg/day for all age and sex groups (Gunderson 1988; Lombardo 1986). Adults had a dieldrin intake of 0.007 µg/kg/day (25–30-year-old males). Dieldrin was found in 15% of the food samples analyzed. These values represent a decrease from the 1980 Total Diet Study. Between 1980 and 1982–1984, daily intakes of dieldrin decreased from 22 ng/kg/day to 8 ng/kg/day for adults (Gunderson 1988). Recently, a Total Diet Study conducted by FDA, found dieldrin in only 6% of the food items analyzed from 1990 (FDA 1991). A daily intake of 0.0016 µg/kg body weight was estimated for 60–65-year-old females, respectively (FDA 1991). The average daily dietary intake of chemical contaminants in food were estimated for 116,957 U.S. adults in 1990 based on annual diet as part of the annual U.S. FDA Total Diet Study (MacIntosh et al. 1996). The estimated mean dietary exposure of dieldrin for 78,882 adult females and 38,075 adult males studied ranged from 0.08 to 0.43 µg/day (mean=0.5 µg/day) and from 0.02 to 4.0 µg/day (mean=0.5 µg/day), respectively. High levels of dietary exposures to dieldrin were estimated to be primarily due to frequent consumption of summer and winter squash, while those with low exposure were dominated by foods that contained residue levels below the limits of detection. During the Total Diet Study conducted by the FDA from November 1993 to June 1994, dieldrin was detected 58 times (concentrations and estimated daily intakes not specified) out of a total of 783 foods sampled (FDA 1995). Assuming that 2 L of water are ingested each day, the average drinking water contribution of dieldrin may range from 0.1 to 0.29 ng/kg/day for a 70 kg adult. These levels are well below the Acceptable Daily Intake (ADI) of 0.1 µg/kg/day recommended by the World Health Organization (WHO) for dieldrin (Geyer et al. 1986). Organohalogen residue levels were monitored from May 1990 to July 1991 in 806 composite milk samples collected from 63 cities within the United States (Trotter and Dickerson 1993). Dieldrin was detected in 172 milk samples ranging from trace amounts to 2 µg/L (detection limit=0.5 µg/L).

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 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

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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).

The widespread use of agricultural pesticides in California has raised concerns about exposures in nearby residential communities, particularly to children (Bradman et al. 1997). To determine the potential exposure to dieldrin, house dust and handwipe samples from children were collected and analyzed from 11 homes, 5 of which had at least 1 farmworker resident. Dieldrin was detected in house dust from one home of a farmworker at a concentration of 0.10 $\mu\text{g/g}$. Dust loading (the fraction dislodgeable by vacuum) of dieldrin was 0.45 $\mu\text{g/m}^2$. These data indicates that the highest chronic daily intake for children would be 1.0×10^{-3} $\mu\text{g/kg/day}$. Nine middle-income households located in the Raleigh-Durham-Chapel Hill area of North Carolina were evaluated for children's pesticide exposure (Lewis et al. 1994). Each house had at least one child in the 6-month to 5-year range. Aldrin was detected in five of the houses, while dieldrin was detected in all nine houses in various matrices (soil samples, dust samples, air samples, etc.). Since dieldrin was detected so often and at higher concentrations, it was studied more intently. The researchers found that the mean concentration of dieldrin was 0.12 $\mu\text{g/g}$ in house dust samples, <0.01 $\mu\text{g/g}$ in child hand rinse samples, 0.01 $\mu\text{g/m}^3$ in air samples taken from the living room, and 0.03 $\mu\text{g/g}$ in play area soil. The estimated exposures of children by respiration and ingestion of house dust ranged from not detectable to 0.13 $\mu\text{g/day}$ and from not detectable to 0.04 $\mu\text{g/day}$, respectively. Judging by these results, it appears that inhalation of indoor air from houses contaminated with aldrin and dieldrin is a major route of child exposure. Due to the greater persistence of dieldrin in the environment, children are expected to have greater exposure to dieldrin than aldrin.

Inhalation of aldrin and dieldrin in outdoor ambient air, however, is not expected to be a significant source of exposure for children. During a study of atmospheric concentrations of chemical contaminants from 1970 to 1972, researchers found that the mean concentrations for aldrin and dieldrin were 0.4 and 1.6 ng/m^3 , respectively (Kutz et al. 1976). Since all but one of their uses were canceled by the EPA in 1974, ambient air concentrations of aldrin and dieldrin are expected to be much lower today. Children living near NPL sites containing high concentrations of aldrin and dieldrin, however, may be exposed to higher than normal atmospheric concentrations. Studies of this nature, however, have not been located. Inhalation exposure may be important during a spill of aldrin or dieldrin before environmental equilibrium is attained. Under these conditions, high concentrations of both compounds would be found in the atmosphere, especially closer to the ground since both compounds are heavier than air. This

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situation, however, is not expected to occur since aldrin and dieldrin are no longer produced or used commercially.

In Streaky Bay, a rural community located on the west coast of South Australia, the contamination of a school by aldrin was studied (Calder et al. 1993). Between August and November 1986, a 0.5% aqueous aldrin emulsion was used within the school as a termiticide. The geometric mean air concentrations of aldrin sampled within the school were 0.09 $\mu\text{g}/\text{m}^3$ in March 1987, 0.11 $\mu\text{g}/\text{m}^3$ in May 1988, 0.05 $\mu\text{g}/\text{m}^3$ in August 1988, and 0.06 $\mu\text{g}/\text{m}^3$ in September 1988. Aldrin contamination was highest in carpet samples with concentrations ranging from 31,600 to 77,000 $\mu\text{g}/100 \text{ cm}^2$. Aldrin is rapidly metabolized to dieldrin and was therefore monitored for in school attendants (Calder et al. 1993). The arithmetic mean concentration of dieldrin in serum samples collected from 138 people was 1.41 ng/mL with a maximum concentration of 9.3 ng/mL in 1987. One year later in 1988, the arithmetic mean concentration of dieldrin decreased to 0.74 ng/mL with a maximum of 2.2 ng/mL.

The FDA Total Diet studies are based on levels found in representative commercially available food products. However, many infants receive human breast milk as a major dietary component rather than milk purchased in grocery stores. Therefore, the daily intake of aldrin and dieldrin by infants may be more closely related to concentrations of dieldrin found in mother's milk. Infants are particularly sensitive to aldrin and dieldrin due to their higher intestinal permeability and immature detoxification system. Dieldrin was found in the breast milk of 80.8% of 1,436 nursing women sampled in 1980, with the greatest percentage (88.9%) in samples collected in the southeastern United States and the lowest percentage from samples collected in the northeast (63.9%) (Savage et al. 1981). The mean fat-adjusted residue level of these samples was 164 ppb. Assuming that milk fat accounts for approximately 3% of whole milk, this would correspond to approximately 5 ppb in whole milk. Of 54 nursing mothers studied in Hawaii (1979–1980), 94% had dieldrin in their milk (Takei et al. 1983). The mean concentration in milk fat was 42 ppb, which would correspond to a concentration of 1.3 ppb in whole milk. Of 57 nursing women sampled in 1973–1974 in Arkansas and Mississippi, 28% had a dieldrin residue level of 4 ppb in their milk (Strassman and Kutz 1977). A level of 0.5 ppb was found in a national survey of the general Canadian population (Davies and Mes 1987).

Several factors may influence the levels of dieldrin found in breast milk. For example, a highly significant ($p < 0.001$) association was reported in women with low levels of dieldrin in breast milk and a history of breast-feeding several children (Ackerman 1980). In addition, women who consume foods lower on the food chain, i.e., vegetarians, had dieldrin levels in their breast milk that were only 1–2% as

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high as the average levels in the United States (Hergenrather et al. 1981). Also, a mother's total body weight may influence the concentration of dieldrin found in breast milk. In a study of Israeli women conducted in 1975, those weighing over 72 kg had significantly lower levels of dieldrin in their breast milk (6 ppb) than those weighing under 63 kg (8.7 ppb) (Polishuk et al. 1977a). This difference was observed despite similar plasma levels of dieldrin in the two groups. A Swedish study found that dieldrin levels in mother's milk decreased from 0.076 $\mu\text{g/g}$ (44 ppb) to 0.010 $\mu\text{g/g}$ (10 ppb) between 1967 and 1984–1985; the use of dieldrin in Sweden was prohibited in 1970 (Norén and Meironyte 2000). A survey of 14 human milk donors whose homes in western Australia had been treated yearly with various pesticides for termite control found dieldrin residues in the milk ranging from 2 to 35 ng/g (2–35 ppb) (mean of 13 ng/g [13 ppb]) (Stacey and Tatum 1985). Milk levels of dieldrin peaked at 7–8 months after house treatment. Three of the 14 houses had recently been treated with aldrin, and the houses of the 11 other donors had been treated with aldrin previously. Dietary intake may have contributed partially to the milk levels since there was not a good correlation between dieldrin and the most recent use of aldrin.

A total of 412 breast milk samples from women in all provinces of Canada were analyzed for organochlorine residues in 1986 (Mes et al. 1993). Dieldrin was detected in 94% of all samples (detection limit=0.009 ng/g) at a mean concentration of 0.46 ng/g (maximum=4.42 ng/g). The study also examined dieldrin concentrations from earlier years. In both 1967 and 1970, the mean concentration of dieldrin in Canadian breast milk samples was 5 ng/g, in 1975, it was 2 ng/g, while in 1982, the concentration dropped to 1 ng/g. Breast milk samples were collected from 23 primiparous mothers and analyzed for their total amount of organochlorine residues from January to November 1992 (Quinsey et al. 1996). The results indicated that the mean daily intake of dieldrin from breast milk would be 0.32 $\mu\text{g/kg}$ body weight/day with a range of 0.06 to 2.24 $\mu\text{g/kg}$ body weight/day. The difference in organochlorine pesticide concentrations in human milk and infant formulas was examined in 1993 (Pico et al. 1995). Human milk samples were obtained from 15 women aged 29–40 living along the Spanish Mediterranean coastal area. The infant formulas analyzed included 11 starting formulas, 11 follow-up formulas, 4 adapted infant formulas, and 17 specialized formulas. Aldrin and dieldrin were not detected in either human milk nor formula samples (detection limit for aldrin=5.1 $\mu\text{g/L}$, detection limit for dieldrin=6.0 $\mu\text{g/L}$).

Studies show that transplacental transfer of aldrin and dieldrin occurs. A study of organochlorine compounds in mothers and fetuses during labor found that dieldrin concentrations in extracted lipids of fetal blood (1.22 ppm) and placenta (0.80 ppm) greatly exceeded those in maternal blood (0.53 ppm) and uterine muscle (0.54 ppm) (Polishuk et al. 1977b). In a study measuring contaminant levels in the cord

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blood of newborn aboriginals and non-aboriginals of the Northwest Territories and Southern Quebec, Canada, mean concentrations of aldrin were found to be 0.01 µg/L in all populations (Van Oostdam et al. 1999). A study of four Iraqi women with no known exposure to organochlorine pesticides found dieldrin levels in the placenta to range from 0.006 to 0.020 mg/kg total tissue weight and average dieldrin levels in their milk to range from 0.007 to 0.023 mg/kg whole milk. However, there was no correlation between the level of dieldrin in the placenta and the level in milk for each individual (Al-Omar et al. 1986).

An FDA Total Diet Study, conducted between 1982 and 1984, found that aldrin intake was <0.001 µg/kg/day for all age and sex groups and that toddlers (2-years-old) had the highest intake levels for dieldrin at 0.016 µg/kg/day, followed by infants at 0.010 µg/kg/day (Gunderson 1988; Lombardo 1986). In 1980 and from 1982 to 1984, daily intakes of dieldrin decreased from 33 to 10 µg/kg/day for infants and from 46 to 16 ng/kg/day for toddlers (Gunderson 1988). The average daily dietary intake for adolescent males (14–16-year-olds) was 0.08 µg/kg/day in 1984. Recently, a Total Diet Study conducted by the FDA found dieldrin residues in only 6% of the food items analyzed from 1990 (FDA 1991). Daily intakes of 0.0014 and 0.0016 µg/kg body weight were estimated for infants 6–11 months old and for 14–16-year-old adolescents in 1990 (FDA 1991). During the Total Diet Study conducted by the FDA from November 1993 to June 1994, dieldrin was detected 58 times (concentrations and estimated daily intakes not specified) out of a total of 783 foods sampled (FDA 1995). The Total Diet Study food list includes many foods eaten by infants and children.

6.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Infants and toddlers are possibly exposed to higher levels of aldrin or dieldrin in the diet than are adults. Table 6-2 is a listing of calculated daily dietary intakes of dieldrin for adults, toddlers, and infants. Infant and toddler dietary intakes decreased significantly from 1978 to 1982. They remained elevated, however, when compared with adult dietary intake.

Higher exposure rates can be expected for large segments of the population residing in homes treated with aldrin or dieldrin for termite control. Measurements of air concentrations in homes 1–10 years after

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Table 6-2. Calculated Dietary Intakes ($\mu\text{g}/\text{kg}$ of Body Weight/Day) of Dieldrin for Three Population Groups^a

Group	1981–1982	1980	1979	1978
Adults	0.016	0.022	0.016	0.017
Infants	0.020	0.033	0.048	0.045
Toddlers	0.023	0.046	0.036	0.039

^aDerived from Gartrell et al. 1986a, 1986b

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pesticide treatment showed dieldrin concentrations ranging from 0.002 to 0.17 ppb in roof voids and from 0.0006 to 0.03 ppb in living rooms and bedrooms and all interior areas (Dobbs and Williams 1983). The indoor air concentrations of aldrin and dieldrin were monitored in the basement, main level, and upstairs area of a treated home from September 1987 to April 1995 (Wallace et al. 1996). In this particular home, aldrin had been poured directly into the foundation blocks during construction. Initially, the aldrin concentrations in air samples taken from the living area and the basement in September 1987 were 300 and 5,000 ng/m³, respectively, while dieldrin concentrations were 7 and 28 ng/m³, respectively. By June 1989, levels of aldrin in the living area and basement were 20 and 300 ng/m³, respectively, while dieldrin concentrations were 5 and 20 ng/m³, respectively. At the end of the study in April 1995, levels of aldrin in the living area and basement were 2 and 12 ng/m³, respectively, while dieldrin concentrations were 3 and 20 ng/m³, respectively. The concentrations of aldrin and dieldrin in air collected outside the home in April 1995 were <0.05 and 0.3 ng/m³, respectively. Eight years after the initial treatment, aldrin and dieldrin were still detected in the living space of the home.

An assessment of the environmental contamination of a residential community built on a thick layer of harbor sludge in the Netherlands, found that the maximal combined daily intake of aldrin, dieldrin, isodrin, and telodrin by soil ingestion, inhalation of contaminated indoor air, and diet exceeded the ADI by a factor of three (Van Wijnen and Stijkel 1988). The concentrations of these compounds were highest in soil samples taken from the top 40 cm. The total indoor air concentrations of the compounds in the living rooms of homes built on contaminated soil were 10 times higher than outdoor air levels (9.9 ng/m³ versus 0.8 ng/m³); levels in the crawl spaces of these homes were 100 times higher (88.7 ng/m³) than outdoor levels although no explanation was given for these elevated levels. Dieldrin concentrations were also elevated in vegetables grown in the soil (up to 40 mg/kg fresh weight) and resulted in a recommendation against the consumption of home-grown vegetables. Dieldrin concentrations were not elevated in drinking water samples in any of the homes tested.

Persons with chronic skin disease may be at increased risk from occupational exposure to pesticides. A formulator with scleroderma had higher blood and tissue levels of dieldrin than did his associates with similar exposures (Hayes 1982). Residents who live near hazardous waste sites that contain aldrin or dieldrin may also have greater exposure to these compounds as a result of contact with contaminated environmental media. Although aldrin is unlikely to persist, dieldrin may enter surface water as a result of surface runoff of contaminated soil. Only limited information is available regarding the extent of contamination at hazardous waste sites and the levels to which individuals may be exposed.

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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 aldrin and dieldrin is available. Where adequate information is not available, ATSDR, in conjunction with the National Toxicology Program (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 aldrin and dieldrin.

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.

6.8.1 Identification of Data Needs

Physical and Chemical Properties. The physical and chemical properties of aldrin and dieldrin are sufficiently well defined to allow assessments of the environmental fate of the compounds to be made (Budavari 1996; Clayton and Clayton 1994; Guerin and Kennedy 1992; Hayes 1982; HSDB 2001a, 2001b; NIOSH 1997; Verschueren 1996). No additional information is needed.

Production, Import/Export, Use, Release, and Disposal. The risk for exposure of the general population to substantial levels of aldrin or dieldrin is quite low. Aldrin and dieldrin have not been produced in the United States since 1974, nor is there any indication that U.S. production of either of these two chemicals will resume (EPA 1990b). Aldrin has not been imported into the United States since 1985 (EPA 1986d). No information was available regarding exports of aldrin or dieldrin, nor was information available regarding the amount of these insecticides currently stockpiled in the United States. Information regarding stockpile levels of aldrin and dieldrin would prove useful.

Currently, all uses of aldrin and dieldrin have been canceled (EPA 1990b). However, due to the persistence of dieldrin in the environment, the likelihood of its bioconcentration, and the former widespread use of both aldrin and dieldrin, these agents are still found at low levels in foods such as root crops and meat and dairy products. Concentrations of dieldrin are significantly higher than aldrin

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residues due to the high rate of conversion of aldrin to dieldrin in the environment and dieldrin's relative stability in environmental matrices.

The soil around dwellings that have been treated with termiticides containing aldrin and dieldrin is the environmental media most likely to be contaminated with significant quantities of aldrin and dieldrin. The air within treated homes may also contain elevated levels of these agents.

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 Toxic Chemical Release Inventory (TRI), which contains this information for 1997, became available in May of 1990. This database will be updated yearly and should provide a list of industrial production facilities and emissions. However, for aldrin and dieldrin, there are no TRI data, indicating that no industrial releases of either of these chemicals were reported for 1997.

Incineration and activated-carbon adsorption have >99% efficiencies as methods for disposing of aldrin or dieldrin (HSDB 2001a, 2001b). However, no information is available regarding the amounts of aldrin or dieldrin disposed of by each method. Additional information on current disposal patterns would prove useful.

Environmental Fate. Aldrin released to surface and shallow subsurface soils partitions to the atmosphere where it is transported (Caro and Taylor 1971; Elgar 1975; McLean et al. 1988). In deeper subsurface soils, aldrin generally is sorbed to soil particulates (McLean et al. 1988); under most environmental conditions, aldrin should not leach to groundwater (McLean et al. 1988). Aldrin is biotransformed to dieldrin in aerobic soils (Gannon and Bigger 1958; Gupta et al. 1979). Additional information is needed on the transformations of aldrin in anaerobic soils and sediments.

Dieldrin sorbs to soils and sediments (Briggs 1981; Cliath and Spencer 1971). The compound also partitions to biota and slowly volatilizes from soils to the atmosphere (Nash 1983). Dieldrin is transported in the particulate phase in surface water runoff (Caro and Taylor 1971; Eye 1968; Hardee et al. 1964) and in the atmosphere (Baldwin et al. 1977). In deep subsurface soils, dieldrin is sorbed to particulates and does not leach to groundwater (Dobbs et al. 1989). The compound is persistent in environmental media, being resistant to biodegradation and abiotic transformation (Gannon and Bigger 1958; Jagnow and Haider 1972). Based on dieldrin's vapor pressure, it will exist in both the vapor and particulate phase in the atmosphere (Grayson and Fosbraey 1982). Vapor-phase dieldrin is expected to

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react with hydroxyl radical; while particulate phase dieldrin will be removed from the atmosphere by wet and dry deposition. Information concerning the relative percentage of dieldrin that will exist in the particulate and vapor-phase in the environment would prove useful in predicting its atmospheric fate.

Bioavailability from Environmental Media. Limited available pharmacokinetic data indicate that the compounds are absorbed by humans following inhalation of contaminated air (Stacey and Tatum 1985). Absorption also occurs following oral and dermal exposures (Feldmann and Maibach 1974; Heath and Vandekar 1964; Hunter and Robinson 1967; Hunter et al. 1969; Iatropoulos et al. 1975). Additional information is needed on the absorption of the compounds following ingestion of contaminated drinking water and soils. This information would be useful in evaluating the importance of various routes of exposure to populations living in the vicinity of hazardous waste sites.

Food Chain Bioaccumulation. Aldrin and dieldrin are bioconcentrated by plants, animals, and aquatic organisms and biomagnified in aquatic and terrestrial food chains (Bhatnagar et al. 1988; Cole et al. 1976; Connell 1989; Donaldson et al. 1999; Metcalf et al. 1973; Sanborn and Yu 1973; Shannon 1977; Travis and Arms 1988). Food chain bioaccumulation appears to be a more important fate process for dieldrin, which is very persistent in nature, than for aldrin, which is rapidly converted to dieldrin (EPA 1980a; Metcalf et al. 1973). No additional information is necessary.

Exposure Levels in Environmental Media. Aldrin and dieldrin have historically been detected in ambient air (Hoff et al. 1996), surface water (EPA 1980a; Stubin et al. 1996), drinking water (EPA 1980a), soils (Eisenreich et al. 1989; Kutz et al. 1976), sediments (Bergersen 1987; Staples et al. 1985), and foods (EPA 1985e; Hundley et al. 1988). Many current monitoring studies indicate that the concentrations of both aldrin and dieldrin in environmental matrices are decreasing (California EPA 1995; MacIntosh et al. 1999; Miller et al. 1992). Aldrin has been identified in at least 207 of the 1,613 hazardous waste sites and dieldrin has been identified in at least 287 of the 1,613 hazardous waste sites that have been proposed for inclusion on the EPA NPL (HazDat 2002). Recent estimates of dietary intake, which is believed to be the most important source of exposure for most members of the general population, are also available (FDA 1991, 1995). More recent monitoring data would be useful in more accurately predicting human exposure.

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Exposure Levels in Humans. The presence of dieldrin in human blood and adipose tissue has been used as an indicator of exposure to aldrin and dieldrin (Brock et al. 1998). The compounds have also been widely detected in human breast milk (Davies and Mes 1987; Quinsey et al. 1996; Savage et al. 1981; Takei et al. 1983). Additional information on the concentration of these compounds in the biological tissue and fluids of populations living in the vicinity of NPL sites would be helpful in assessing the extent to which these populations have been exposed to these compounds.

Exposures of Children. With the detection of dieldrin in drinking water (Kolpin et al. 1997), studies that detail the exposure of infants fed formula prepared from tap water would prove helpful. More data are needed to properly assess aldrin and dieldrin exposure to children who live, play, or attend school near NPL sites and farmlands that have been treated with these pesticides. Information regarding the number of houses in the United States that have been treated with aldrin and dieldrin formulations in the past would be useful in determining the number of children that would be potentially exposed today. The stability of these compounds, especially dieldrin, suggests the possibility that they may be brought home by farm workers who work on farmlands previously treated with these compounds. More exposure studies that monitor aldrin and dieldrin exposure to children of farm workers would be extremely valuable.

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 aldrin and dieldrin were located. These substances are not currently any of the compounds for which subregistries have been established in the National Exposure Registry. These substances 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 these substances.

Information is particularly needed on the size of the populations potentially exposed to aldrin and dieldrin through contact with contaminated media in the vicinity of hazardous waste sites. The development of an exposure registry would provide a useful reference tool in assessing exposure levels and frequencies. It would also facilitate the conduct of epidemiological or health studies to assess any adverse health effects resulting from exposure to aldrin and/or dieldrin. In addition, a registry developed on the basis of exposure sources would allow an assessment of the variations in exposure levels from one source to

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another and the effect of geographical, seasonal, and regulatory action on the level of exposure within a certain source. These assessments, in turn, would provide a better understanding of the needs for research or data acquisition on the current exposure levels.

6.8.2 Ongoing Studies

A pilot project is under way in South Dakota to identify types and levels of pesticide residues in breast milk of South Dakota residents and to evaluate the effect of diet and maternal weight change on proximate composition and pesticide excretion levels in milk. The project will also estimate pesticide loading in breast-fed and non-breast-fed infants. To date, trace amounts of dieldrin (>0.001 ppm) have been detected in human milk samples. Levels of dieldrin appear to decrease from week 1 to week 7 postpartum.

Remedial investigations and feasibility studies conducted at the NPL sites contaminated with aldrin and dieldrin will add to the available database on exposure levels in environmental media and in humans and will contribute information for exposure registries. Investigations at the sites will also increase the current knowledge regarding the transport and transformation of aldrin and dieldrin at hazardous sites. No other long-term research studies regarding the environment fate and transport of aldrin and dieldrin or the occupational and general population exposure to these compounds were identified.

The Federal Research in Progress (FEDRIP 2001) database and the Current Research and Information System database funded by the U.S. Department of Agriculture (CRIS/USDA 2001) provide additional information obtainable from a few ongoing studies that may fill in some of the data needs identified in Section 6.8.1. These studies are summarized in Table 6-3.

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Table 6-3. Ongoing Studies on the Potential for Human Exposure to Aldrin and Dieldrin

Investigator	Affiliation	Research description	Sponsor	Source
Childress C	Department of Interior, U.S. Geological Survey, Water Resources Division, North-Central North Carolina	There is no long-term regional water-quality monitoring data for many of the streams and tributaries to the drinking water supplies of the Research Triangle area of North Carolina. Many of these streams continue to receive a complex combination of treated industrial and municipal waste, in addition to nonpoint urban and agricultural runoff. This study is aimed at documenting the spatial differences in regional surface-water quality. The study will test for dieldrin concentrations.	U.S. Geological Survey	FEDRIP 2001
Morlock S	Department of Interior, U.S. Geological Survey, Water Resources Division, Northwest Indiana	To build an extensive database of major contaminants in the most contaminated tributaries of Lake Michigan and to assess the mobility of these contaminants. This information can be used to estimate the loads of these contaminants to Lake Michigan. The contaminants to be studied are 100 polychlorinated biphenyl (PCB) congeners, dieldrin, and chlordane.	U.S. Geological Survey	FEDRIP 2001

