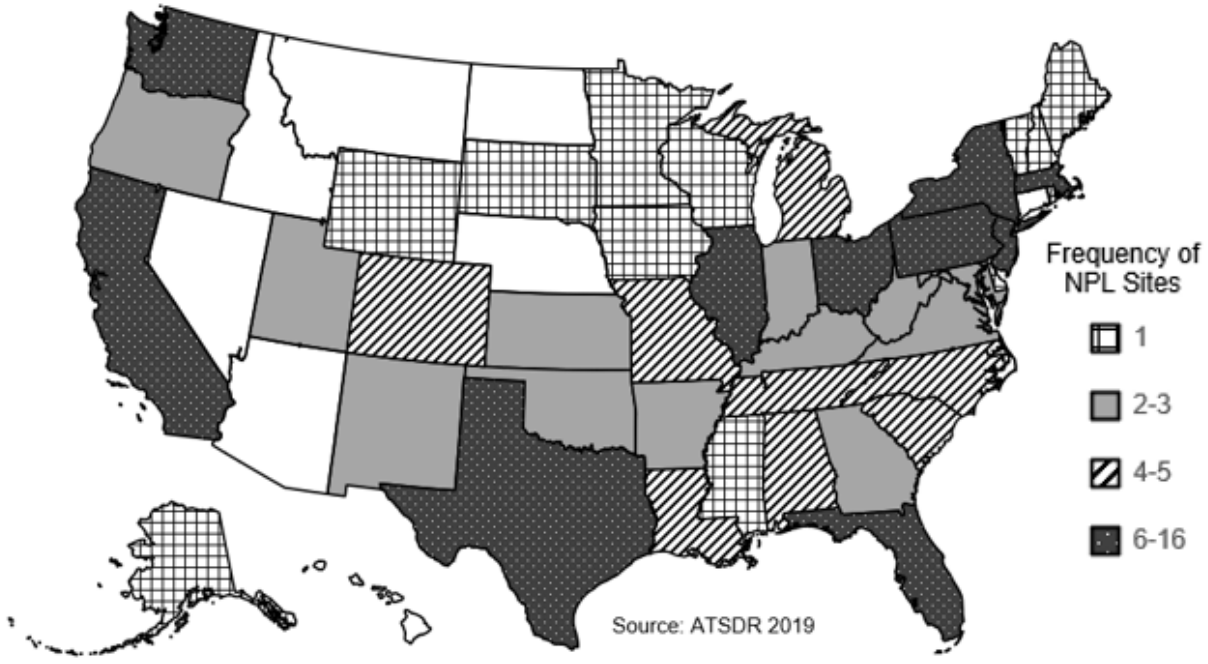
CHAPTER 5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

Endrin, endrin aldehyde, or endrin ketone has been identified in at least 176 of the 1,867 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2019). However, the number of sites in which endrin, endrin aldehyde, or endrin ketone has been evaluated is not known. The number of sites in each state is shown in Figure 5-1. Of these sites, 175 are located within the United States and 1 is located in the Virgin Islands (not shown).

Figure 5-1. Number of NPL Sites with Endrin, Endrin Aldehyde, or Endrin Ketone Contamination



- The most likely route of exposure to endrin for the general population is via ingestion of endrin residues on imported food items. Endrin is no longer used in the United States.
- Environmental exposure is expected to be low. Exposure to small amounts may occur following unregistered use, inappropriate disposal, or at hazardous waste sites.
- Currently, there are no significant releases of endrin into the environment, but endrin persists in sorbed forms to sediments and soil from previous use.
- The estimated half-life of endrin in soil is approximately 14 years. It is not expected to migrate from soil to groundwater.

- Endrin transformation products including endrin ketone, endrin aldehyde, and endrin alcohol, can be detected in plants grown in soils treated at least 16 years prior to planting.

5.2 PRODUCTION, IMPORT/EXPORT, USE, AND DISPOSAL

5.2.1 Production

No information is available in the TRI database on facilities that manufacture or process endrin because this chemical is not required to be reported under Section 313 of the Emergency Planning and Community Right-to-Know Act (Title III of the Superfund Amendments and Reauthorization Act of 1986) (EPA 2005).

Endrin is a stereoisomer of dieldrin produced by the reaction of vinyl chloride and hexachlorocyclopentadiene to yield a product that is then dehydrochlorinated and condensed with cyclopentadiene to produce isodrin. This intermediate is then epoxidized with peracetic or perbenzoic acid to yield endrin. An alternative production method involves condensation of hexachlorocyclopentadiene with acetylene to yield the intermediate for condensation with cyclopentadiene (EPA 1985b; IARC 1974).

Endrin is no longer manufactured in the United States. Velsicol Chemical Company, Memphis, Tennessee, was the producer of endrin until the final voluntary cancellation of registration with the Office of Pesticide Programs in 1991 (EPA 1984a, 1985a, 1986a, 1993; USDA 1993). It is estimated that 2.345 million kg (5.1–9.9 million pounds) of endrin were sold in the United States in 1962, while <450,000 kg (990,000 pounds) were produced in 1971 (IARC 1974). More recent estimates of domestic production of endrin could not be found (NLM 2020). As with many toxic chemicals, information on production or use of pesticides is often proprietary, and quantitative estimates of production of endrin are virtually impossible to obtain (Bason and Colborn 1992). Chemical manufacturers in the United States, however, can legally produce pesticides for export that are currently banned or not registered for use in the United States (FASE 1996).

Endrin aldehyde and endrin ketone were never commercial products but occurred as impurities of endrin or as degradation products (EPA 1985b; IARC 1974; SRI 1987). While commercial preparations of solid endrin were typically 95–98% pure, the following chemicals (in addition to endrin aldehyde and endrin ketone) have been found as trace impurities: aldrin, dieldrin, isodrin, heptachloronorbornadiene, and heptachloronorborene (NLM 2020). The active ingredient would often be mixed with one or more

organic solvents for application in a liquid form. Carriers included xylene, hexane, and cyclohexane (NLM 2020; Zabik et al. 1971).

5.2.2 Import/Export

Data on historic imports and exports of endrin are sparse. The most recent data that could be located indicate that about 21,000 kg (46,000 pounds) of endrin were imported into the United States in 1972 (IARC 1974). No information on export volumes of endrin was located. However, the Foundation for Advancements in Science and Education reported that almost 75% of the 750,000 tons of pesticides that the United States exported from 1992 to 1994 lacked chemical-specific information (FASE 1996). Many of the exported pesticides were organochlorine pesticides that had been banned for use in the United States.

5.2.3 Use

Endrin was first used as an insecticide, rodenticide, and avicide beginning in 1951 to control cutworms, voles, grasshoppers, borers, and other pests on cotton, sugarcane, tobacco, apple orchards, and grain (EPA 1979a; NLM 2020). It was also used as an avicide on enclosed bird perches (EPA 1985c). Unlike aldrin/dieldrin, with which it has many chemical similarities, endrin apparently was never used extensively for termite-proofing or other applications in urban areas (Blus et al. 1989; NLM 2020). Endrin's toxicity to nontarget populations of raptors (birds of prey) and migratory birds was a major reason for its cancellation as a pesticide agent (Blus et al. 1989; EPA 1979b; USDA 1993). Except for use as an avicide on bird perches, which was canceled in 1991, all other uses of endrin in the United States were voluntarily canceled by the manufacturer in 1986 (EPA 1984a, 1985a, 1986a, 1993; USDA 1993). It has been estimated that 6,250 kg (13,780 pounds) of endrin were used annually in the United States prior to 1983 (EPA 1986c). Since endrin may still be used as a pesticide agent in foreign countries, residues on imported food items are still of some concern (FDA 1990, 1991, 1992; Hundley et al. 1988). Both the EPA and the Food and Drug Administration (FDA) revoked all food tolerances for endrin in 1993 (USDA 1993).

5.2.4 Disposal

Because endrin and endrin aldehyde are listed as hazardous substances, disposal of wastes containing these compounds is controlled by a number of federal regulations (see Chapter 7). Land disposal restrictions apply to wastes containing endrin or endrin aldehyde (EPA 1987a). Chemical treatment

(reductive dechlorination) or incineration are possible disposal methods (IRPTC 1985; NLM 2020). Past disposal methods included land disposal (EPA 1987b; Sittig 1980). In general, disposal methods for endrin residues or endrin-containing wastes are similar to those for wastes containing aldrin/dieldrin (NLM 2020). No information was found in the available literature on regulations or methods for the disposal of endrin ketone.

No information was found in the available literature on the amounts of endrin, endrin aldehyde, or endrin ketone disposed of in the United States by any method.

5.3 RELEASES TO THE ENVIRONMENT

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

Because virtually all uses of endrin in the United States were voluntarily canceled by 1986 (EPA 1984a, 1985a, 1986a, 1993; USDA 1993), releases to the environment of endrin, or endrin aldehyde and endrin ketone, which occur as impurities or degradation products of endrin, have decreased dramatically over the last decade.

5.3.1 Air

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

In the past, emissions from endrin production and processing facilities and agricultural applications were primary sources of releases of endrin to the atmosphere. During the period when endrin was extensively used in agriculture, 33% of the applied endrin was found to volatilize within 11 days, after which time further evaporation ceased (Nash 1983).

There is also a potential for atmospheric release of endrin, endrin aldehyde, and endrin ketone from hazardous waste sites.

5.3.2 Water

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

In the past, endrin could have been released to surface water from manufacturing and processing facilities. No information on direct discharges or loadings of endrin into surface water was found. Based on amounts measured in rainfall at various stations in Canada, loading estimates for endrin and a number of other organochlorine pesticides have been attempted for portions of the Great Lakes basin (Strachan 1988). The sources for such loadings to receiving waters are not clear, but would likely involve in-place contaminants related to endrin's past uses as a pesticide agent. A 1995 study conducted in Oklahoma indicates that in some areas of the United States, endrin can be released to surface water from farmland soils that have been treated with endrin in the past (Petty et al. 1995).

There is also a potential for release of endrin, endrin aldehyde, and endrin ketone to water from hazardous waste sites.

5.3.3 Soil

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

Past use of endrin as an agricultural pesticide has been the principal source of its release to soils or aquatic sediments. There is also a potential for release of endrin, endrin aldehyde, and endrin ketone to soils and sediments from hazardous waste sites.

5.4 ENVIRONMENTAL FATE

5.4.1 Transport and Partitioning

Endrin is extremely persistent when released to the soil. It adsorbs strongly to soil particles and tends to be immobile, based on an estimated K_{oc} of 34,000 (Kenaga 1980; Swann et al. 1983). Endrin on soil may be transported to surface water via runoff from rain or irrigation. Since endrin in solid form is hydrophobic and sorbs strongly to soil particles, migration into groundwater would not generally be expected from normal agricultural application. In laboratory studies, endrin was found to be almost completely adsorbed to samples of sandy loam and organic soil (Sharom et al. 1980a). In sandy soil, only 13.6% of the endrin was leached from the soil after 10 successive 200 mL water rinses. In organic soil, only 1.5% of the endrin was leached from the soil after 10 successive 200 mL water rinses. The mobility factors calculated for the sandy soil and organic soil were 0.52 and 0.040, respectively. Only dieldrin, leptophos, and *p,p'*-DDT were less mobile than endrin in the two soil types.

However, endrin has been detected in groundwater, suggesting that leaching may be possible in some soils under certain conditions (CSWRCB 1986; EPA 1988). Furthermore, because endrin formulations in solvent carriers such as xylene or hexane were also commonly used, endrin could move into groundwater from spills of such formulations. Similarly, migration to groundwater might also occur at waste sites where endrin residues become mixed with organic solvents (Jaquess et al. 1989).

Despite endrin's low vapor pressure of 2.0×10^{-7} mm Hg (EPA 1981), initial volatilization of 20–30% after agricultural application to soil has been reported to be rapid (Nash 1983). Within 11 days, however, further volatilization was no longer detected (Nash 1983). Unlike some other chlorinated pesticides, endrin volatilization was not enhanced after a rainfall. Small amounts of endrin in soil may also be transported to the air by dust particles.

The presence of significant concentrations of endrin transformation products (including endrin ketone, endrin aldehyde, and endrin alcohol) in a variety of plants grown in soil treated with endrin for periods as long as 16 years prior to planting (Nash et al. 1972; Nash and Harris 1973) indicates that there may be significant uptake of endrin and/or its transformation products by plants from endrin-treated soil.

Because of its high log K_{oc} and log K_{ow} values (4.53 and 5.34–5.6, respectively; see Table 4-2), when released to water, endrin strongly adsorbs to sediment (Kenaga 1980; Swann et al. 1983) and bioconcentrates significantly in aquatic organisms (ASTER 1995; EPA 1980; Metcalf et al. 1973).

Typical bioconcentration factors (BCFs) for freshwater and marine organisms range from 80 to 49,000 (Table 5-1). Biomagnification of endrin with increasing trophic level is not expected to be significant (Leblanc 1995). Metcalf et al. (1973) reported a ratio of biomagnification through the aquatic food chain to bioconcentration by direct uptake from water to be 2 for endrin compared to 2.50 for DDT. These authors used a model laboratory aquatic ecosystem containing algae (*Oedogonium cardiacurn*), snails (*Physa* sp.), water fleas (*Daphnia magna*), mosquito larvae (*Culex pipiens quinquefasciatus*), and mosquito fish (*Gambusia afinis*).

Table 5-1. Bioconcentration Data for Endrin

Species common name (scientific name)	Exposure type	Duration (days)	BCF[a]	Reference
Freshwater				
Algae (*Microcystis aeruginosa*)	–	7	200	EPA 1980
Algae (*Anabaena cylindrica*)	–	7	222	EPA 1980
Algae (*Scenedesmus quadricauda*)	–	7	156	EPA 1980
Algae (*Oedogonium* sp.)	–	7	140	EPA 1980
Water flea (*Daphnia magna*)	S	1	2,600	Metcalf et al. 1973
Mosquito (*Culex pipiens quinquefasciata*)	S	1	2,100	Metcalf et al. 1973
Stonefly (*Pteronarcys dorsata*)	F	28	1,000	Anderson and Defoe 1980
Pouch snail (*Physa* sp.)	S	33	49,000	Metcalf et al. 1973
Mussels (mixed species)	–	21	3,000	Jarvinen and Tyo 1978
Channel catfish (*Ictalurua punctatus*)	–	41–55	2,000	EPA 1980
Flagfish (*Jordanella floridae*)	–	65	15,000	Hermanutz 1978
Flagfish (*J. floridae*)	F	15	7,000	Hermanutz et al. 1985
Fathead minnow (*Pimephales promelas*)	F	47	10,000	EPA 1980
Fathead minnow (*P. promelas*)	–	56–300	7,000	Jarvinen and Tyo 1978
Fathead minnow (*P. promelas*)	F	2–304	80b	Veith and Kosian 1983
Black bullhead (*Ictalurus melas*)	F	4	3,700	Anderson and Defoe 1980
Black bullhead (*I. melas*)	F	7	6,200	Anderson and Defoe 1980
Saltwater				
Grass shrimp (*Palaemonetes pugio*)	F	145	1,600	Tyler-Schroeder 1979
American oyster (*Crassostrea virginica*)	F	2	1,670	Mason and Rowe 1976
American oyster (*C. virginica*)	F	7	2,780	Mason and Rowe 1976
Sheepshead minnow, embryo-juveniles (*Cyprinodon variegatus*)	–	33	4,800	EPA 1980
Sheepshead minnow (*C. variegatus*)	–	141–161	6,400	Hansen et al. 1977
Spot (*Leiostomos xanthurus*)	–	5–8 months	1,450	EPA 1980

[a]BCF listed is the highest bioconcentration factor (BCF) value reported in the cited reference.
[b]Calculated quantitative structure-activity relationship (QSAR) value.

F = flow-through exposure system; S = static system

Based on its very small calculated Henry's law constant of $4.0x10^{-7}$–$5.4x10^{-7}$ atm-m^3/mol (see Table 4-2) and its strong adsorption to sediment particles, endrin would be expected to partition very little from water into air (Thomas 1990). The half-life for volatilization of endrin from a model river 1 meter deep, flowing 1 meter per second, with a wind speed of 3 meters per second, was estimated to be 9.6 days, whereas a half-life of >4 years was estimated for volatilization of endrin from a model pond (Howard 1991). Adsorption of endrin to sediment may reduce the rate of volatilization from water.

In air, endrin is expected to be associated primarily with particulate matter based on its low vapor pressure and high K_{oc} (Kenaga 1980). However, small amounts of endrin in the atmosphere may exist in the vapor phase (Eisenreich et al. 1981). Because of its low solubility (200 µg/L, see Table 4-2), endrin would not be expected to be removed significantly from the atmosphere by wet deposition. Particle-adsorbed endrin will be removed from the atmosphere by both wet and dry deposition. Endrin was found in 5% of 450 wet deposition (rain/snow) samples collected between 1986 and 1991 in the Great Lakes area, at volume weighted mean concentrations ranging from 0.02 to 0.98 ng/L (ppt) (Chan et al. 1994).

No studies on the environmental transport and partitioning of endrin aldehyde could be found in the available literature. Values of the estimated log K_{ow} for endrin aldehyde vary widely, ranging from 3.1 to 5.6 (see Table 4-2). Based on the lowest estimated log K_{ow}, the K_{oc} value for endrin aldehyde can be estimated to be approximately 1,000 (Lyman 1990), indicating a low mobility in soil (Swann et al. 1983). Using the higher estimated values of log K_{ow} (4.7–5.6), the K_{oc} value for endrin aldehyde can be estimated to range from 8,500 to 380,000 (Lyman 1990), indicating that this compound will be virtually immobile in most soils (Swann et al. 1983). Because of its low vapor pressure of $2.0x10^{-7}$ mmHg and Henry's Law constant ranging from $2x10^{-9}$ to $3.7x10^{-8}$ atm-m^3/mol (see Table 4-2), endrin aldehyde would not be expected to volatilize significantly from soil or water (Eisenreich et al. 1981; Thomas 1990). Any endrin aldehyde in air should exist predominantly in the adsorbed phase (Eisenreich et al. 1981). Atmospheric endrin aldehyde will be transported to soil and surface water via wet and dry deposition of associated particles. In water, adsorption to sediments and bioconcentration are likely to be significant partitioning processes. Based on the lowest estimated value of 3.1 for log K_{ow} (see Table 4-2), the BCF value for endrin aldehyde can be estimated to be only 86 (Veith et al. 1979), indicating little tendency to bioconcentrate in aquatic organisms. Using the higher estimates of 4.7–5.6 for log K_{ow} (see Table 4-2), BCF values for endrin aldehyde are estimated to range from to 2,000 to 11,000 (Veith et al. 1979), indicating a much higher tendency for bioconcentration.

No studies on the environmental transport or partitioning of endrin ketone could be found in the available literature, and only limited information was found on estimated values of physical and chemical properties. The very low estimated value of 2.02×10^{-8} atm-m^3/mole for Henry's Law constant for endrin ketone (see Table 4-2) indicates that this compound will not volatilize from water. Based on the estimated log K_{ow} of 4.99 (see Table 4-2), the BCF value for endrin ketone can be estimated to be 3,500 (Veith et al. 1979), indicating that endrin ketone may be removed from water via bioconcentration in aquatic organisms. Also based on an estimated log K_{ow} of 4.99, the K_{oc} value for endrin ketone can be estimated to range from 5,500 to 90,000 (Lyman 1990), indicating that this compound will be virtually immobile in soil and sediments (Swann et al. 1983).

5.4.2 Transformation and Degradation

Air. Field studies on the transformation of endrin in the atmosphere were not located in the available literature. Photochemical isomerization of endrin, primarily to the pentacyclic ketone commonly called delta ketoendrin or endrin ketone, was observed after exposure of thin layers of solid endrin on glass to sunlight (Burton and Pollard 1974). Minor amounts of endrin aldehyde were also formed in this reaction. Results of seasonal studies indicated that this isomerization would proceed with a half-life (first-order kinetics) of 5–9 days in intense summer sunlight, with complete conversion to the pentacyclic ketone in 15–19 days. Knoevenagel and Himmelreich (1976) reported that photodegradation of solid endrin in the laboratory proceeded with a half-life (first-order kinetics) of 20–40 hours. In laboratory studies conducted by Zabik et al. (1971) on endrin formulations in hexane and cyclohexane (similar to those commonly used for pesticide applications), endrin was found to undergo photolytic dechlorination when exposed to ultraviolet radiation, yielding a pentachlorinated half-cage ketone as the major product. This degradation product was also detected in environmental samples. Endrin may also be transformed by heat in the atmosphere, yielding primarily the pentacylic ketone and endrin aldehyde (EPA 1979c; Phillips et al. 1962). Endrin may also react with photochemically generated hydroxyl radicals in air, with a predicted half-life (first-order kinetics) ranging from 1.45 hours (Howard 1991) to 1.8 days (EPA 1995b). No information could be found on the products of this reaction. The reaction of endrin with ozone in air is not significant. The predicted first-order rate constant for this reaction is 3.6×10^{-20} cm^3/molecule-second, corresponding to a half-life of 320 days (EPA 1995b).

Endrin aldehyde may react with photochemically generated hydroxyl radicals in the atmosphere, with an estimated overall first-order rate constant of $106x10^{-12}$ cm^3/molecule-second, which corresponds to a half-life of 3.6 hours, assuming a 24-hour concentration of hydroxyl radicals of $0.5x10^6$ molecules/cm^3 (EPA 1995b). Endrin ketone may react with photochemically generated hydroxyl radicals in the atmosphere, with an estimated overall first-order rate constant of $10.8x10^{-12}$ cm^3/molecule-set, which corresponds to a half-life of 1.5 days, assuming a 24-hour concentration of hydroxyl radicals of $0.5x10^6$ molecules/cm^3 (EPA 1995b). No other information could be found in the available literature on the transformation and degradation of endrin aldehyde or endrin ketone in air.

Water. Laboratory studies of the fate of endrin in water samples suggest a significant degree of stability, although there is evidence of varying degrees of biodegradation in some systems. Endrin was among the more stable of 12 insecticides incubated in water collected from the drainage canal of a vegetable-growing site near Toronto, with about 80% of endrin remaining in the natural water after incubation for 16 weeks (Sharom et al. 1980b). There was little indication of chemical degradation of endrin in these studies. Studies in which sealed water samples from the Little Miami River were exposed to sunlight and artificial fluorescent light showed no measurable degradation of endrin over an 8-week period (Eichelberger and Lichtenberg 1971). However, microorganisms in fish pond water and algae from a fish pond were able to metabolize endrin (Patil et al. 1972). In the case of the algae, the metabolite was 12-ketoendrin. The rate of metabolism was 35% for the water sample and 24% for the algal culture in 1 month. Using the static culture procedure, Tabak et al. (1981) found no biodegradation of endrin in domestic waste water samples.

Based on laboratory experiments on solid endrin (Burton and Pollard 1974) and on endrin in organic solvents (Zabik et al. 1971), it is likely that endrin released to surface water will undergo photo-isomerization to endrin ketone, with minor amounts of endrin aldehyde also being formed. Under real world conditions, endrin released to surface waters would not be expected to biodegrade or hydrolyze to any significant extent (Eichelberger and Lichtenberg 1971; EPA 1979c). Endrin is very resistant to hydrolysis, with an estimated half-life (first-order kinetics) of >4 years (EPA 1979c). The predominant removal of endrin from water by photodegradation and sorption to suspended particulates or sediments is consistent with the observed low incidence of detected endrin in ambient surface waters based on analyses of EPA National Urban Runoff Program (Cole et al. 1984) and STORET (Staples et al. 1985) data.

Little information could be found in the available literature on the transformation and degradation of endrin aldehyde in water. Neither hydrolysis nor oxidation (via peroxy radicals or singlet oxygen) are expected to be significant in aquatic systems (EPA 1981). By analogy to endrin, the hydrolysis half-life (first-order kinetics) of endrin aldehyde in water is probably >4 years (EPA 1979c). No information could be found on the biodegradation of endrin aldehyde in aquatic systems.

No information could be found in the available literature on the transformation and degradation of endrin ketone in water.

Sediment and Soil. Biodegradation does not appear to be a significant degradation process for endrin in soils. The actual measurement of biodegradation of endrin under field conditions on well-drained agricultural soil indicates a biodegradation half-disappearance time of approximately 14 years (Nash and Woolson 1967), suggesting that endrin is resistant to biodegradation in soils under natural conditions. In this study, 41% of the initial endrin applied to an agricultural field was present in the soil after 14 years.

Laboratory studies indicate that endrin can be biodegraded in various soils under various conditions; however, caution should be exercised in extrapolating laboratory results to field conditions. Twenty different isolates of soil organisms belonging to several different species (five identified, four unidentified) were found to biodegrade endrin in the laboratory under aerobic conditions (Patil et al. 1970). The study revealed endrin as one of the more easily biodegradable insecticides, while lindane, for example, was not degraded by any of the 20 isolates. In contrast, Bartha et al. (1967) found no biodegradation of endrin, but they used rather insensitive analytical techniques compensated for by high endrin concentrations (0.25 ppm) that would not occur in normal agricultural practice. Nitrification was enhanced by endrin in this experiment. Endrin was also biodegraded to four unidentified metabolites in laboratory microcosms using flooded rice soils (Gowda and Sethunathan 1976). The most rapid degradation was seen in the saline acid sulfate soil, pokkali, followed by alluvial and laterite soils, where endrin concentrations dropped 10–20-fold in 55 days. Sandy soils were least active and reduced endrin concentration only by about 40% in 55 days. The addition of organic matter, such as rice straw, approximately doubled the rate of biodegradation. Half-disappearance times of endrin in soils ranged from <20 days under optimal conditions to about 80 days under less-favorable conditions. A degradation half-life (first-order kinetics) of 26–32 weeks was reported for endrin (initial concentration approximately 1.6–2.0 ppm) in a clay soil under controlled, aerobic environmental conditions (30°C; soil water content 10–33%), with slower degradation observed in soils with the lowest moisture content (Ghadiri et al. 1995). First-order rate equations best described the degradation. Virtually complete anaerobic

biodegradation of endrin in laboratory microcosms within 4 days has been reported; however, the researchers caution that under natural conditions, redox environments in many soils will not be suitable for anaerobic degradation, and that endrin residues sorbed to soil particles would often be rendered unavailable to bacteria (Maule et al. 1987).

In combination, losses from volatilization, photodegradation (Burton and Pollard 1974; EPA 1985b; Knoevenagel and Himmelreich 1976; Zabik et al. 1971), and heat transformation (primarily to endrin ketone, with minor amounts of endrin aldehyde) (EPA 1979c; Phillips et al. 1962) are likely to account for a rapid decrease in endrin residues on soil or plant surfaces exposed to bright sunlight. Studies have also been conducted indicating significant concentrations of endrin transformation products (including endrin ketone, endrin aldehyde, and endrin alcohol) in plants grown in endrin-treated soil (Nash et al. 1972; Nash and Harris 1973).

Little information could be found in the available literature on the transformation and degradation of endrin aldehyde in sediment and soil. By analogy to aquatic systems, neither hydrolysis nor oxidation (via peroxy radicals or singlet oxygen) would be expected to be significant transformation processes. No information could be found on the biodegradation of endrin aldehyde in sediment or soil.

No information could be found in the available literature on the transformation and degradation of endrin ketone in sediment and soil.

5.5 LEVELS IN THE ENVIRONMENT

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

Table 5-2 shows the lowest limit of detections that are achieved by analytical analysis in environmental media.

Table 5-2. Lowest Limit of Detection Based on Standards[a]

Media	Detection limit	Reference
Air	No data	Harkov 1986; Ligocki and Pankow 1985
Water	0.006 ppb	EPA 1982
Soil/sediment	2–3 ppb	Carey et al. 1976
Blood serum	5.09–7.8 ng/g lipid	CDC 2018

[a]Detection limits based on using appropriate preparation and analytics. These limits may not be possible in all situations.

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

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

Medium	Median[a]	Geometric mean[a]	Geometric standard deviation[a]	Number of quantitative measurements	NPL sites
Endrin					
Water (ppb)	0.355	0.651	62.2	16	13
Soil (ppb)	1,810	1,160	53.7	25	24
Air (ppbv)	0.0013	0.0024	61	7	4
Endrin aldehyde					
Water (ppb)	0.17	0.099	29	6	5
Soil (ppb)	6.05	86.8	10.4	20	17
Air (ppbv)			No data		
Endrin ketone					
Water (ppb)	0.465	0.622	18.2	12	8
Soil (ppb)	52	202	27.7	27	20
Air (ppbv)			No data		

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

5.5.1 Air

Endrin is relatively nonvolatile with a vapor pressure of 2.0×10^{-7} mmHg (EPA 1981; Worthing and Walker 1983). Despite its low volatility, initial loss of agriculturally applied endrin through volatilization was found to be comparable to more volatile pesticides (Nash 1983). Endrin was detected in seven of

132 ambient air samples collected by the EPA Great Lakes National Program in 2018; reported concentrations were 0.143–0.688 pg/m^3 (NWQMC 2020). In 2 of 29 dry fall material samples collected, reported concentrations were 0.774 and 0.847 pg/m^3 (NWQMC 2020).

Limited information was found on atmospheric concentrations of endrin between 1970 and the mid-1980s, prior to cancellation of virtually all uses (EPA 1984a, 1985a, 1986a, 1993; USDA 1993). The data were insufficient to identify any trends. The mean and maximum airborne concentrations of endrin in the United States in 1970–1971 were reported to be 0.2 and 19.2 ng/m^3, respectively (Lee 1977). For that same time period, mean airborne concentrations at suburban sites near Jackson, Mississippi, and Columbia, South Carolina, were reported to be 0.1 and 0.2 ng/m^3, respectively (Bidleman 1981; Kutz et al. 1976). Endrin was not detected at Boston, Massachusetts, suburban sites (Bidleman 1981). A survey of airborne contaminants in the Great Lakes area in 1981 did not detect endrin (Eisenreich et al. 1981).

Atmospheric concentrations of endrin in the vicinity of manufacturing facilities were higher than those found in non-source areas of the United States. Eight hundred meters from two formulation plants in Arkansas and 275 meters from one formulation plant in Tennessee, mean airborne concentrations of endrin were reported to be 3.3 and 3.5 ng/m^3, respectively, during 1970–1972 (Lewis and Lee 1976). Endrin was also detected in air in industrial or source-dominated regions of the Mississippi Delta in 1972–1974, and in Tennessee in 1971 (EPA 1985b).

Endrin may also be found in atmospheric precipitation. In an analysis of pesticides in rainfall from four stations in Canada in 1984, detectable concentrations of endrin were found at each site (Strachan 1988). There was a noticeable pattern of decline in detections within the summer season (May–August). In studies in the Great Lakes area, endrin was found in 5% of 450 wet deposition (rain/snow) samples collected between 1986 and 1991, at volume weighted mean concentrations ranging from 0.02 to 0.98 ng/L (ppt) (Chan et al. 1994).

No information was found in the available literature on concentrations of endrin aldehyde or endrin ketone in ambient outdoor air or in indoor air. In addition, no information was available on occupational exposures to these chemicals.

5.5.2 Water

Endrin was not reported above the lower quantification limit of 6.0–60 ng/L (ppt) in over 3,600 ambient surface water data points compiled for 2016–2020 from EPA STOrage and RETrieval (STORET) and National Water Information System (NWIS) databases (NWQMC 2020). Endrin aldehyde was also not reported above the lower quantification limit of 5.0–68 ng/L (ppt) in approximately 2,700 surface water data points reported (NWQMC 2020). Unlike DDT, chlordane, aldrin/dieldrin, and a variety of other chlorinated pesticides, endrin was never used extensively in urban areas. This is reflected in the results from EPA's Nationwide Urban Runoff Program, which showed no detections in 86 high-flow water samples from 51 urbanized watersheds from 19 cities (Cole et al. 1984). Endrin was not detected (detection limit 49 ng/L [0.045 ppb]) in surface water from the Yakima River Basin, Washington (Foster et al. 1993). However, in 1991–1992, endrin was detected in first flush (first 20 minutes) stormwater runoff samples at four of six sites in Louisville, Kentucky, at levels that exceeded U.S. federal criteria (0.003 µg/L, [ppb]) (Marsh 1993). Maximum, minimum, and mean concentrations at the four sites were 0.03–0.05, 0.02–0.04, and 0.03–0.04 (ppb), respectively. Endrin was not detected (detection limit not specified) in 3-hour composite samples of stormwater runoff from any of the six sites.

Endrin is rarely detected in drinking water and any trace amounts of endrin that might be encountered in raw drinking water supplies will likely be removed in the treatment systems used by most communities. In 1966 and 1967, when the use of endrin was not restricted, endrin was detected in 5 of 67 raw water samples from the Mississippi and Missouri Rivers (Schafer et al. 1969). When endrin use was substantially restricted, an Iowa study of 33 community water supplies using surface water found no detectable concentrations of endrin in the distribution systems (IDNR 1987). In an extensive water quality monitoring program conducted by the California Department of Health Services, endrin was detected (detection limit not specified) in only 2 of 5,109 public drinking water sources sampled from 1984 to 1992, at mean and maximum concentrations of 0.06 and 0.10 ppb, respectively (Storm 1994). Concentrations did not exceed the maximum concentration level of 0.2 ppb. Endrin was not detected (detection limit not specified) in 32 samples each of raw water and highly treated reclaimed waste water undergoing evaluation as a possible supplement to raw water sources in San Diego, California (De Peyster et al. 1993). Recent data concerning endrin concentrations in drinking water were not available.

Detections of endrin in groundwater are also rare except from wells near hazardous waste sites. The EPA Pesticides in Groundwater Data Base (EPA 1988) contains groundwater data collected with good quality assurance/quality control (QA/QC) provisions from areas with significant agricultural land uses as well as

from urban areas. Analysis of these data indicated there were only two wells with detectable levels of endrin within the entire United States. A detection occurred in a well in California (concentration not reported) where point source problems or spills were deemed the likely sources. Endrin contamination found in an Illinois well at an average concentration of 0.02 ppb was considered likely to have resulted from ordinary agricultural uses. In a groundwater contamination study of California's 58 counties, in which over 50 pesticides were evaluated from both point and nonpoint sources, endrin was detected in only one sample (CSWRCB 1986). Endrin was detected at 0.9% of 178 Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) sites and 1.3% of 156 Resource Conservation and Recovery Act (RCRA) sites sampled; however, endrin concentrations were not reported (Plumb 1987). Endrin was not reported above the lower quantification limit of 10–28 ng/L (ppt) in 480 groundwater data points compiled for 2016–2020 from EPA STORET and NWIS databases (NWQMC 2020). Endrin aldehyde was also not reported above the lower quantification limit of 21–28 ng/L (ppt) in 360 groundwater data points reported (NWQMC 2020).

No information was found in the available literature on levels of endrin ketone in surface or groundwater.

5.5.3 Sediment and Soil

In general, endrin was found infrequently and at relatively low levels in both urban and cropland soils in the United States. Endrin was detected in only 10 of 1,483 cropland soil samples in 1972 at concentrations up to 2.13 ppm (detection limit of 0.01 ppm) (Carey et al. 1979). These studies were part of the National Soils Monitoring Program carried out by EPA and the U.S. Department of Agriculture (USDA) under the National Pesticide Monitoring Program, which covered a total of 1,533 sampling sites in 37 states. Endrin detections were documented in the following states: Alabama, Arkansas, Georgia, Illinois, Louisiana, Nebraska, New York, North Carolina, and South Dakota, as well as at sites from one or more of the mid-Atlantic states of Delaware, Maryland, and New Jersey. Endrin was also detected at a level of 0.017 ppm at a single site in California in a study that targeted rice-growing cropland soils in Arkansas, California, Louisiana, Mississippi, and Texas (Carey et al. 1980). Endrin was not detected in urban soils from 13 of 14 U.S. cities included in a 1970 study of pesticide residues in urban soils (25–30 soil sampling sites were used for each of the urban areas) (Carey et al. 1976). The only detection was at a single site near Memphis, Tennessee, where the Velsicol Chemical Company (which produced endrin at that time) is located. The reported concentration for this site was 0.07 ppm; the mean concentration for all 28 Memphis sites was <0.01 ppm. Endrin and endrin aldehyde was detected in 6 of 119 soil data

points compiled from EPA STORET; concentrations were 22.9–42 µg/kg (ppb) for endrin and 17.5–26.6 µg/kg (ppb) for endrin aldehyde (NWQMC 2020).

Relatively little literature was identified concerning the analysis of endrin in aquatic sediments. The available data indicate that, historically, sediment concentrations of endrin have been very low. In a study of sediment contaminants in Casco Bay, Maine, endrin was found at concentrations near or below the method detection limit (<0.25 ppb) (Kennicutt et al. 1994). In the National Surface Water Monitoring Program conducted from 1976 to 1980, endrin was detected in 1.3% of the sediment samples analyzed (detection limit not reported), with a maximum concentration of 2.9 ppb (Carey and Kutz 1985). In a study by Ford and Hill (1991) to evaluate organochlorine pesticide residues in sediments and aquatic animals in the vicinity of the Yazoo National Wildlife Refuge in the Mississippi Delta, a region that has experienced very high usage of pesticide agents, detectable levels of endrin were not found in sediments (detection limit 0.01 ppm [10 ppb]). Similarly, endrin was not detected in sediment or pore water samples (detection limits 0.49 and 0.01 ppm [490 and 10 ppb], respectively) from 18 mosquito control impoundments in St. Lucie County, Florida, where organochlorine pesticides had been heavily used through the early 1960s (Parkinson et al. 1993). Endrin was detected in 42 of 973 sediment data points compiled for 2016 to 2020 from EPA STORET; concentrations were 0.52–16.24 µg/kg (ppb) (NWQMC 2020). Endrin aldehyde was not detected (detection limit 0.017–2.9 µg/kg [ppb]) in 678 sediment data points reported (NWQMC 2020).

No information was found in the available literature on levels of endrin aldehyde in soil or endrin ketone in sediment or soil.

5.5.4 Other Media

Endrin had been found in many foods, but current levels appear to be very low and not of concern for human health. The FDA has concluded that endrin is no longer present in the environment to the extent that it may be contaminating food or feed at levels of regulatory concern (USDA 1993). An FDA survey of pesticide residues in samples of domestic and imported food and feed commodities from Fiscal Year (FY) 1982 to 1986 lists endrin levels for specific food items up to 0.50 ppm (500 ppb) (Hundley et al. 1988). This study was conducted by surveillance sampling with follow-up compliance sampling for sources of foodstuffs where the concentrations in surveillance samples violated EPA tolerance levels. In surveillance sampling, endrin was detected in 0.05% (3 of 6,391 samples) and 1.5% (183 of 12,044 samples) of domestic and imported foods, respectively. The incidence of violative surveillance

samples (endrin residues ≥0.05 ppm [50 ppb]) was higher for imported foods (0.1%; 12 violations) than for food items from domestic sources (0.02%; 1 violation). In follow-up compliance monitoring of 1,239 samples of imported foods, endrin was found in 11 samples (0.9%); 2 of these samples (0.2%) were violative. In imported foods, endrin was detected in mung beans, cucumbers, pickling cucumbers, cantaloupe, acorn squash, cabocha squash, Italian squash, summer squash, and yellow squash (Hundley et al. 1988). In another study of pesticide residues in food conducted in 10 states between 1988 and 1989, endrin was not detected in any of the 13,980 samples analyzed in 1988. In 1989, the detection frequencies for endrin and endrin ketone were 0.084 and 0.007%, respectively, for the 13,085 samples analyzed (Minyard and Roberts 1991). Endrin was detected each year in regulatory monitoring of domestic and imported foods conducted by the FDA from 1989 to 1994 as part of its Pesticide Residue Monitoring Program (FDA 1990, 1991, 1992, 1993, 1994, 1995). Concentrations were not reported; however, <1% of the surveillance samples had any pesticide residue levels that were above established tolerances. In the most recent FDA survey of pesticide residues in U.S. domestic food, endrin was not detected in the 1,931 food samples (FDA 2017a).

Endrin was also detected in the FDA Total Diet Studies in 1987, 1988, 1989, and 1991, but not in 1990 (FDA 1988, 1989, 1990, 1991, 1992). Overall, in 234 ready-to-eat foods tested 37 times each as part of the FDA Total Diet Studies from 1982 to 1991, endrin was found only 26 times at an average concentration of 0.0027 µg /g (2.7 ppb) in nine different foods: broccoli, cantaloupe, collards, cucumbers, onion rings, dill pickles, pumpkin pie, summer squash, and winter squash (Kan-Do Office 1995). Concentrations ranged from 0.0011 µg/g (1.1 ppb) (broccoli) to 0.0041 µg/g (4.1 ppb) (summer squash). In a summary of 1985–1991 FDA pesticide residue findings, endrin was not reported in >10,000 surveillance samples of domestic and imported foods that may be eaten by infants or children, or in >4,000 analyses of Total Diet Study foods eaten by infants and children (Yess et al. 1993). In FDA Total Diet Studies between 1991–1993 and 2003–2004, endrin was detected in seven food items (canned evaporated milk, cantaloupe, cucumber, summer squash, winter squash, radish, and cheese pizza); it was only detected in 10 of the 296 samples (0.03%) with levels ranging from 0. 2 to 2.0 ppb (FDA 2006). In 2017, endrin was not detected in whole milk samples (limit of detection of 1.5 ppb) (FDA 2017b). In an overall summary for the 5-year period 1986–1991, average dietary intakes of endrin for eight age/sex groups (6–11-month-old infants, 2-year-old children, 14–16-year-old males and females, 25–30-year-old males and females, and 60–65-year-old males and females) were all estimated to be <0.000l µg/kg body weight/day (FDA 1993).

Other studies further indicate that the occurrence of endrin in the U.S. food supply is very low. In a 1990–1991 FDA survey of pesticide residues in milk representing most of the U.S. supply consumed in metropolitan areas, endrin was detected at trace levels (0.0005–0.001 ppm [0.5–1.0 ppb]) in only 2 of 806 composite samples (one sample each from Atlanta, Georgia and Dover, Delaware) (Trotter and Dickerson 1993). In another statistically based FDA study in 1992–1993, endrin was not found as a violative residue in any of 710 domestic or 949 imported pear samples (Roy et al. 1995). Endrin was not reported among the pesticides detected in a 1994 FDA survey of pesticide levels in 160 samples of catfish, crayfish, shrimp, trout, salmon, oysters, and various other species from important aquaculture areas of the United States (FDA 1995). Comparable results were found in similar studies conducted by the FDA in 1990–1993 (FDA 1995). A food basket survey patterned after the FDA approach that was conducted in San Antonio, Texas did not find detectable concentrations of endrin (detection limit 0.050 ppm [50 ppb]) in 6,970 produce items (Schattenberg and Hsu 1992).

Because of the persistence of endrin in the environment and its potential to bioconcentrate significantly in aquatic organisms, there has been continued concern over the levels of endrin in fish and shellfish. This concern, however, appears to be limited primarily to specific sites where endrin was used heavily in agriculture or was discharged by industrial plants. In 1963, at the height of agricultural endrin use, endrin levels in catfish poisoned by endrin exceeded 4 ppm (4,000 ppb) during a fish kill (Mount and Putnicki 1966). Endrin was detected in two species of commercial Penaeus shrimp collected at 21 of 31 stations in the Calcasieu River Estuary in Louisiana, a Gulf Coast estuary receiving both industrial discharges and urban and agricultural runoff (Murray and Beck 1990). The maximum, mean, and median concentrations of endrin reported were 9.47, 1.07, and 0.25 ppm (9,470, 1,070, and 250 ppb), respectively. Several other national studies, however, indicated that contaminated fish or shellfish currently are not a likely source of potentially high human exposure to endrin. In the National Contaminant Biomonitoring Program, maximum endrin concentrations in whole fish from around the United States for the periods 1976–1777, 1978–1979, 1980–1981, and 1984 were 0.40, 0.11, 0.30, and 0.22 ppm (400, 110, 300, and 220 ppb), respectively. Corresponding geometric mean concentrations were ≤0.01 ppm (10 ppb) (Schmitt et al. 1985, 1990). The percentage of stations where detectable endrin residues were present also showed a relatively steady decline from 47.2% in 1976–1977 to 28% in 1984. The maximum concentration of 0.22 ppm (220 ppb) in 1984 was recorded near Memphis in the vicinity of the Velsicol Chemical Company. In portions of the Mississippi Delta within or bordering the Yazoo National Wildlife Refuge, endrin was found at the 0.01 ppm (10 ppb) detection limit in whole-body tissue samples from such rough fish as carp, smallmouth buffalo, bowfin, and spotted gar collected in 1988 (Ford and Hill 1991). In the 1986 National Study of Chemical Residues in Fish conducted by the EPA, endrin was detected in fish

tissue samples at 11% of the 362 sites surveyed. The maximum, mean, and median concentrations of endrin reported were 0.162 ppm, 0.002 ppm, and not detected (<0.0025 ppm) (162, 2, and <2.5 ppb), respectively (EPA 1992a).

Endrin concentrations also have been monitored in several studies in the Great Lakes region. Endrin was detected in eight fish species from three Great Lakes-influenced rivers in Michigan at average concentrations ranging from not detected (detection limit not specified) to 8.03 ppb wet weight (Giesy et al. 1994). Average concentrations exceeded 1.5 ppb for samples from only 2 of 23 species/site combinations and were <0.5 ppb for samples from 17 of 23 species/site combinations. Endrin was detected (detection limit 0.02 ppm [20 ppb] wet weight) in 5 of 10 samples of lake trout (mean concentration 0.03+0.01 ppm [30+10 ppb]) collected in Lake Michigan in 1982 (Miller 1993). It was not detected in 10 samples of lake trout collected in Lake Superior or in 18 samples of chinook salmon collected in Lake Michigan. Endrin was not detected (detection limit 2 ng/g [ppb] wet weight) in 16 skinless fillets of both rainbow trout (*Oncorhyncus mykiss*) and black bullheads (*Ameiurus melas*) cultivated for 6 and 3.5 months, respectively, in Lake Ontario waters (Buttner et al. 1995). Endrin also was not detected (detection limit not specified) in samples of whole Zebra mussels (*Dreissena polymorpha*) from populations infesting two power-generating stations in Lake Erie (Doherty et al. 1993).

In the 2009 National Study of Chemical Residues in Lake Fish Tissues, endrin was detected in 3 of 486 samples from predator fish with a maximum concentration of 12 ppb (EPA 2009). It was also detected in 14 of 395 samples of bottom dweller fish with a maximum concentration of 20 ppb (EPA 2009); only 1 sample exceeded the minimum level of 10 ppb.

Endrin has been detected in several marine fish species in regional or state monitoring studies. From 1990 to 1993, endrin was found in 40 of 47 whole or fillet samples of red drum (*Sciaenops ocellatus*) at two of four sites along the South Carolina coast, at mean concentrations of 5.61+8.94 and 0.65+3.67 ppb wet weight (Mathews 1994). In this same study, endrin was found in 33 of 74 flounder (*Paralichthys lethostigma*) samples and in 19 of 58 seatrout (*Cynoscion nebulosus*) samples at only one coastal site, at mean concentrations of 0.14+0.81 and 2.68+11.13 ppb, respectively. Endrin was detected in all of 10 liver tissue samples from cod (*Gadus morhua*) in the Northwest Atlantic at a mean concentration of 9 ppb (range, 5–19 ppb), but not in muscle or ovary samples (Hellou et al. 1993).

There may be a potential for contamination of fish and wildlife in the vicinity of hazardous waste sites.

5.6 GENERAL POPULATION EXPOSURE

Endrin is no longer registered for use in the United States. Consequently, the current potential for exposure of the general population to endrin appears to be very limited and will likely continue to diminish even more over time. Members of the general population may be exposed to very low levels of endrin through ingestion of contaminated foodstuffs, particularly those that are imported from areas where endrin is still being used. However, FDA concluded that endrin is no longer present enough in the environment to contaminate food or feed at levels of regulatory concern (USDA 1993). Several studies indicate that human exposures are far below the levels of concern for human health. Based on results of FDA Total Diet Studies conducted from 1978 to 1982, estimated average dietary intakes were <0.001 µg/kg (ppb) body weight/day for infants and toddlers for all 5 years (Gartrell et al. 1986). However, actual intakes must have been lower than these estimates because the reported average dietary intakes were based on the mean concentration of the positive samples only. A report summarizing the FDA Total Diet Studies from April 1982 to April 1984 indicated an estimated daily intake for 6–11-month-old infants of 0.0003 µg/kg (ppb)body weight/day for that period, with estimated daily intakes for 14–16-year-old males and 60–65-year-old females essentially zero (Gunderson 1988). Endrin intakes, in µg/kg (ppb) body weight/day, estimated from the Total Diet Study analyses were <0.0001, <0.0001, and 0.0001 in FY 1989 for 6–11-month-old infants, 14–16-year-old males, and 60–65-year-old females, respectively (FDA 1990), and <0.000l in FY 1991 for all age categories (FDA 1992). Estimated endrin intakes were not reported for FY 1990 (FDA 1991). In an overall summary of FDA Total Diet Studies for the 5-year period 1986–1991, average dietary intakes of endrin for eight age/sex groups (6–11-month-old infants, 2-year-old children, 14–16-year-old males and females, 25–30-year-old males and females, and 60–65-year-old males and females) were all estimated to be <0.0001 µg/kg (ppb) body weight per day (FDA 1993). In Canada, where endrin was registered for use from 1954 to 1990, a dietary intake study estimated the adult annual intake of endrin at approximately 32 µg (0.001 µg/kg [ppb] body weight/day) (Davies 1988).

The National Health and Nutrition Examination Survey (NHANES) uses biomonitoring to provide estimates of exposure to the civilian U.S. population. Chemicals and their metabolites are measured in subsets of participants aged 6–59 years old, meant to be a representative sample of the general U.S. population. Serum endrin values were surveyed during the years 2001–2004; endrin was pooled with other organochlorines in subsequent NHANES surveys (CDC 2019). As shown in Table 5-4, endrin was below the limit of detection (LOD) in most samples. The LODs for survey years 2001–2002 and 2003–

2004 are 5.09 and 7.8 ng/g lipid, respectively. The analytical method used for the analysis was gas chromatography with high-resolution mass spectrometry.

Although endrin bioaccumulates significantly in aquatic organisms (ASTER 1995; EPA 1980; Metcalf et al. 1973), studies indicate that endrin levels in fish and shellfish in the United States are not of concern for human health (EPA 1992b; Ford and Hill 1991; Murray and Beck 1990; Schmitt et al. 1985, 1990). Dietary exposures to endrin from domestic fish were estimated from 1984 to 1988 FDA surveillance data to be $1.7x10^{-5}$ µg/kg (ppb) body weight/day (Ahmed et al. 1993). As of 1995, there were no fish consumption advisories for endrin in effect in the United States (EPA 1995c).

In the 1982 National Human Adipose Tissue Survey, endrin was not detected in adipose tissues from the general U.S. population (EPA 1986b). Endrin also was not detected in adipose breast tissue from breast cancer patients (n=5) or controls (n=5) in the United States (Djordjevic et al. 1994). A 1984 study based on autopsied adipose tissue from 141 cadavers from six Canadian Great Lakes municipalities showed no detectable concentrations of endrin (detection limit 2.4 ppb) (Williams et al. 1988). In a 1990–1991 survey, only very low levels of endrin (average concentration 3.27 ng/g (ppb); range 0.23–8.56 ng/g [ppb] lipid) were found in adipose tissue samples from 3 of 41 residents of British Columbia, Canada, where endrin was registered for use from 1954 to 1990 (Teshke et al. 1993).

Table 5-4. Serum Endrin Levels (Lipid-Adjusted) in the NHANES U.S. Population[a]

	Survey years[b]	Geometric mean (95% CI) (ng/g lipid)	Selected percentiles (95% confidence interval) (ng/g lipid)				Sample size
			50th	75th	90th	95th	
Total	2001–2002	Not calculated[c]	<LOD	<LOD	<LOD	5.10 (<LOD–5.40)	2,187
	2003–2004	Not calculated	<LOD	<LOD	<LOD	LOD	1,825
Age group							
12–19 years	2001–2002	Not calculated	<LOD	<LOD	5.20 (<LOD–5.50)	5.60 (5.40–5.70)	730
	2003–2004	Not calculated	<LOD	<LOD	<LOD	<LOD	539
≥20 years	2001–2002	Not calculated	<LOD	<LOD	<LOD	<LOD	1,457
	2003–2004	Not calculated	<LOD	<LOD	<LOD	<LOD	1,286
Sex							
Males	2001–2002	Not calculated	<LOD	<LOD	<LOD	5.20 (<LOD–5.40)	1,022
	2003–2004	Not calculated	<LOD	<LOD	<LOD	<LOD	885
Females	2001–2002	Not calculated	<LOD	<LOD	<LOD	<LOD	1,165
	2003–2004	Not calculated	<LOD	<LOD	<LOD	<LOD	940
Race/ethnicity							
Mexican Americans	2001–2002	Not calculated	<LOD	<LOD	<LOD	5.30 (<LOD–6.50)	547
	2003–2004	Not calculated	<LOD	<LOD	<LOD	<LOD	433
Non-Hispanic blacks	2001–2002	Not calculated	<LOD	<LOD	<LOD	5.40 (<LOD–6.30)	487
	2003–2004	Not calculated	<LOD	<LOD	<LOD	<LOD	446
Non-Hispanic whites	2001–2002	Not calculated	<LOD	<LOD	<LOD	5.10 (<LOD–5.40)	1,000
	2003–2004	Not calculated	<LOD	<LOD	<LOD	<LOD	831

[a]Serum endrin levels not adjusted for lipid levels are available at https://www.cdc.gov/exposurereport/pdf/FourthReport_UpdatedTables_Volume2_Jan2019-508.pdf.
[b]The limit of detection for survey years 2001–2002 and 2003–2004 were 5.09 and 7.8 ng/g lipid, respectively. Serum endrin levels were not measured after NHANES 2004.
[c]Not calculated: proportion of results below LOD was too high to provide valid results.

LOD = limit of detection; NHANES = National Health and Nutrition Examination Survey

Source: CDC 2019

Endrin has been detected in the milk of lactating women living outside the United States; however, no data from the United States could be located. Data from other countries indicate that there is some correlation between the levels of endrin used in, or transported to, an area and concentrations found in breast milk. Endrin was not detected in breast milk samples from a remote area of Papua, New Guinea (Spicer and Kereu 1993). In a recent investigation of Inuit exposure to organochlorine pesticides through the aquatic food chain in Arctic Quebec, endrin was detected in only 1 of 107 breast milk samples from Inuit women, at a concentration of <8 ng/g (ppb) in milk fat, and in none of 50 samples from southern Quebec Caucasian women (Dewailly et al. 1993). In France, where endrin has not been used for over 20 years, endrin was detected in 8 of 20 human milk samples collected 20–90 days after parturition. Concentrations ranged from 0.02 to 0.84 ppm (20–840 ppb) in milk fat, with a mean concentration of 0.06 ppm (60 ppb) (Bordet et al. 1993). Higher levels of endrin were found in human milk in a study conducted in Jordan, where endrin was widely used over the previous 40 years (Alawi et al. 1992). In this study, endrin was detected in samples from 3 of 15 donors at concentrations ranging from 0.26 to 6.24 ppm (260–6,240 ppb) in milk fat. The median and maximum daily intakes of endrin for breastfed infants were estimated to be 1.55 and 12.70 mg/kg (ppm) (1,550 and 12,700 ppb) body weight, respectively. The relevance of these findings to the U.S. population is unclear.

5.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Endrin has not been registered for use in the United States since voluntary cancellation of its final use as an avicide on bird perches in 1991 (USDA 1993). All other uses of endrin were voluntarily canceled by 1986 (EPA 1984a, 1985a, 1986a, 1993; USDA 1993). Therefore, there are currently no population groups exposed to high levels of endrin associated with its application as a pesticide agent. Populations exposed to higher than background concentrations of endrin, endrin aldehyde, or endrin ketone include those living near hazardous waste sites where these compounds are present. Skin contact with or ingestion of endrin-contaminated soil may be an important source of exposure for children living near such hazardous waste sites. In addition, groundwater may be a source of exposure to endrin for adults and children if they consume drinking water from contaminated wells.

Although all uses of endrin in the United States were canceled by 1991 (EPA 1984a, 1985a, 1986a, 1993; USDA 1993), occupational exposures to endrin, endrin aldehyde, and endrin ketone may occur among workers involved in the handling and treatment of materials at hazardous waste sites, and among agricultural workers at sites formerly treated with endrin. No information was found in the available literature on current occupational exposures. In the past, exposures of agricultural workers were

significant. Seasonal agricultural workers dusting potatoes with 1% endrin dust were calculated to be exposed to a dermal dose of 2.0 mg/kg (ppm) body weight/day and an inhalation dose of 0.04 mg/kg (ppm) body weight/day at a time when agricultural use of endrin was near its peak (Wolfe et al. 1963).

Occupational exposure to endrin was not evaluated during the National Occupational Exposure Survey (NOES) conducted from 1981 to 1983 or its predecessor, the National Occupational Hazard Survey (NOHS) conducted from 1972 to 1974. The surveys conducted by the National Institute for Occupational Safety and Health (NIOSH) were designed to provide data necessary to describe potential exposure agents and profile health and safety programs in United States workplaces.