6. POTENTIAL FOR HUMAN EXPOSURE

6.1 OVERVIEW

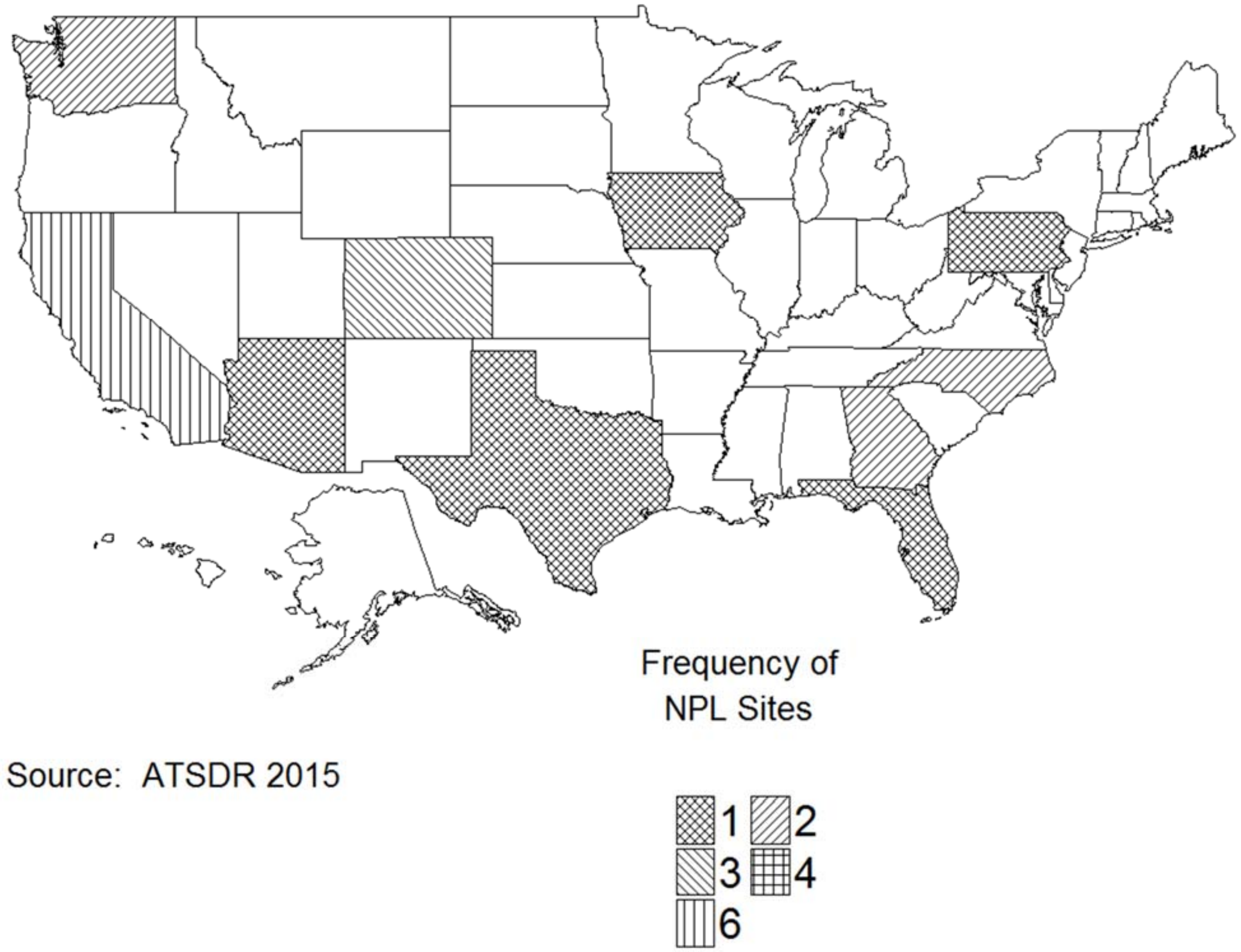
Parathion has been identified in at least 20 of the 1,832 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2015b). However, the number of sites evaluated for parathion is not known. The frequency of these sites can be seen in Figure 6-1. Detections of parathion in environmental media at NPL sites are summarized in Table 6-1.

6.2 RELEASES TO THE ENVIRONMENT

Parathion has not been manufactured, processed, or used in the U.S. mainly since 2000; so, it is likely that no significant releases to the environment have occurred in several years. The state of California reported that small quantities of parathion were used by a few growers each year from 2002 to 2014. Reported parathion usage was reduced from a high of 1,542 pounds applied to 713 acres in 2006 to 22 pounds applied to 1 acres in 2014. The latest use was for landscape maintenance, plants in containers, and structural pest control. The reports noted that the pesticide might actually have been another pesticide, but misreported each year as parathion (CalEPA 2015). Parathion was also formerly released into the atmosphere by human activities associated with its production and use as an insecticide for agricultural purposes.

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

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



Source: ATSDR 2015

Table 6-1. Parathion Levels in Water, Soil, and Air of National Priorities List (NPL) Sites

Medium	Median[a]	Geometric mean[a]	Geometric standard deviation	Number of quantitative measurements	NPL sites
Water (ppm)	0.0004	0.00147	9.57	3	2
Soil (ppm)	18.5	24	104	14	6
Air (µg/m^3)	No data	No data	No data	No data	No data

[a]Concentrations found in Agency for Toxic Substances and Disease Registry (ATSDR) site documents from 1981 to 2015 for 1,832 NPL sites (ATSDR 2015). Maximum concentrations were abstracted for types of environmental media for which exposure is likely. The number of concentrations approximates the number of pathways the contaminant was found in. Pathways do not necessarily involve exposure or levels of concern.

During its use as a broad spectrum insecticide in the United States, parathion directly entered the environment from point sources. These are associated with specific points of release, and from nonpoint sources, which are diffuse and widely dispersed, such as runoff to streams, seepage to groundwater, and deposition of pesticides from the atmosphere (USGS 2007). In order to reduce exposure to children and others, a 1991 agreement began that made parathion a restricted use organophosphate insecticide. In September 2000, most manufacturers agreed to cancel the registration of manufacturing use products. This was followed by the termination of registration of those end use products effective December 31, 2002. Most legal use of parathion products was cancelled as of October 31, 2003 (EPA 2000).

The Drexel Chemical Company was still authorized to continue manufacturing parathion through 2003 and still had four products actively registered under FIFRA. On March 16, 2005, Drexel requested the cancellation of these four remaining products and this became effective on December 13, 2006 (EPA 2006b). There are no known natural sources of parathion.

6.2.1 Air

Estimated releases of 5 pounds (~1.36×10^{-3} metric tons) of parathion to the atmosphere from two domestic manufacturing and processing facilities in 2014 (TRI14 2015). These releases are shown in Table 6-2. The two facilities identified in the TRI are hazardous waste incinerators and it is likely that these reportings occurred as a result of incinerating old unused stockpiles of product.

Parathion was also formerly released into the atmosphere by human activities associated with its production and use as an insecticide for agricultural purposes. One study assessed the rate at which parathion applied to soil or mixed with water (simulating wet agricultural soil and a retention pond, respectively) would volatilize to the surrounding air. The measured rates were low and similar to those predicted from the Henry's law constant, and were 0.8% d^{-1} for wet surface soil and 0.003 hr^{-1} of the surface water surface water concentration (Sanders and Seiber 1984). These values might overestimate what would occur in the environment where degradation and uptake by biota are in competition.

6.2.2 Water

No release of parathion to surface water or publically owned treatment works (POTWs) from two domestic manufacturing and processing facility in 2014 was reported from facilities required to report to the TRI (TRI14 2015). These releases are listed in Table 6-2.

Table 6-2. Releases to the Environment from Facilities that Produce, Process, or Use Parathion[a]

		Reported amounts released in pounds per year[b]							
							Total release		
State[c]	RF[d]	Air[e]	Water[f]	UI[g]	Land[h]	Other[i]	On-site[j]	Off-site[k]	On- and off-site
TX	2	5	0	0	0	No data	5	No data	5
Total	2	5	0	0	0	0	5	0	5

[a] The TRI data should be used with caution since only certain types of facilities are required to report. This is not an exhaustive list. Data are rounded to nearest whole number.
[b] Data in TRI are maximum amounts released by each facility.
[c] Post office state abbreviations are used.
[d] Number of reporting facilities.
[e] The sum of fugitive and point source releases are included in releases to air by a given facility.
[f] Surface water discharges, waste water treatment-(metals only), and publicly owned treatment works (POTWs) (metal and metal compounds).
[g] Class I wells, Class II-V wells, and underground injection.
[h] Resource Conservation and Recovery Act (RCRA) subtitle C landfills; other onsite landfills, land treatment, surface impoundments, other land disposal, other landfills.
[i] Storage only, solidification/stabilization (metals only), other off-site management, transfers to waste broker for disposal, unknown
[j] The sum of all releases of the chemical to air, land, water, and underground injection wells.
[k] Total amount of chemical transferred off-site, including to POTWs.

RF = reporting facilities; UI = underground injection

Source: TRI14 2015 (Data are from 2014)

Parathion was formerly released into water directly from point source discharges, from drift during pesticide applications, and from nonpoint-source runoff from agricultural areas.

6.2.3 Soil

No releases of parathion to soils from two domestic manufacturing and processing facilities in 2014 were reported to the TRI (TRI14 2015). These releases are summarized in Table 6-2.

Parathion was formerly released into soils primarily from its registered use on various agricultural crops. In agricultural areas, parathion may also have been transferred to aquatic sediments (Wan et al. 1994).

Since parathion undergoes various reduction and degradation reactions in the course of time ranging from hours to months, these loadings to soils and sediments were temporary phenomena.

6.3 ENVIRONMENTAL FATE

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

6.3.1 Transport and Partitioning

Based on the vapor pressure of parathion (see Table 4-2) and the organic liquids with which it typically is mixed, parathion released to the atmosphere via agricultural spraying would be expected to exist in both the vapor and particulate phases (Eisenreich et al. 1981).

Parathion released to water from both point and nonpoint sources may be sorbed to soils and sediments, or accumulated in aquatic organisms. While volatilization of parathion may not be expected to be significant based upon the Henry’s law constant (see Table 4-2), it can be an important transport process from water surfaces. Sanders and Seiber (1984) measured a volatilization rate of 30×10^{-4}/hour from water for parathion, corresponding to a half-life of 9.6 days. A laboratory experiment determined that the volatilization half-life for parathion from water 4.5 cm deep was 14 and 9.3 days from unstirred and stirred solutions, respectively, at 24°C. This is equivalent to 311 and 206 days, respectively, from water 1 meter deep (Chiou et al. 1980). In another laboratory experiment designed to simulate an evaporation pond, 0.3% of parathion volatilized after 1 day (Sanders and Seiber 1983). Parathion released to water may also be adsorbed highly by soils and sediments based on its organic carbon partition coefficient (log K_{oc}) values measured to range from 2.50 to 4.20 for varying types of soil and sediment (Mingelgrin and Gerstl 1983).

The adsorption of parathion to soils and sediment attenuates the rate of volatilization and its mobility. Adsorption to suspended solids and sediment in the water column also affects its susceptibility to photolysis, its bioavailability for aquatic organisms, and its biodegradation (Schuurmann et al. 2006).

Parathion does not significantly bioaccumulate in aquatic organisms. A compilation of bioconcentration factors (BCFs) obtained for various freshwater fish and other species is presented in Table 6-3. The BCF values range from 63 to 462. Tadpoles, which are generally resistant to cholinesterase inhibitors such as parathion and are therefore suspected that they might accumulate the pesticide, had a reported BCF that averaged 64 (Hall and Kolbe 1980). This residue level in tadpoles is consistent with BCFs for vertebrate species.

Parathion may be metabolized in fish and other aquatic species after prolonged exposure. Some biotransformation pathways have been recognized. Sheepshead minnows (*Cyprinidon variegatus*) metabolized parathion to paraoxon, a toxic product, through catalysis by P450-dependent monooxygenation (James 1994). Parathion may be metabolized in fish and other aquatic species after prolonged exposure. Some biotransformation pathways have been recognized, such as metabolism of parathion to aminoparathion, with subsequent transformation to *p*-nitrophenol by fish microbes; see Figure 6-2 (EPA 1977b). Sheepshead minnows (*Cyprinidon variegatus*) metabolized parathion to paraoxon, a toxic product, through catalysis by P450-dependent monooxygenation (James 1994). Hydrolysis of parathion has been demonstrated in several shrimp and crayfish species (James 1994).

Parathion released to soil partitions to the atmosphere through volatilization, to surface water via runoff, and to groundwater as a result of leaching. Volatilization of parathion from moist and dry soils is not expected to be a significant transport process based upon the Henry's law constant and vapor pressure (see Table 4-2). The vapor loss rate of technical-grade parathion from a non-absorbing glass surface is 0.210 $\mu g/cm^2/hour$ (Spencer et al. 1979). Vapor losses from parathion-incorporated soil are expected to be much lower due to binding by the soil. Simulations of parathion losses from dry soil resulted in only 0.1–0.3% losses in 30 days when incorporated into 10 cm of soil (Jury et al. 1984). Volatilization in a laboratory environmental chamber designed to simulate a soil pit resulted in 0.8% volatilization in 1 day from wet soil and approximately an order of magnitude less from dry soil (Sanders and Seiber 1984).

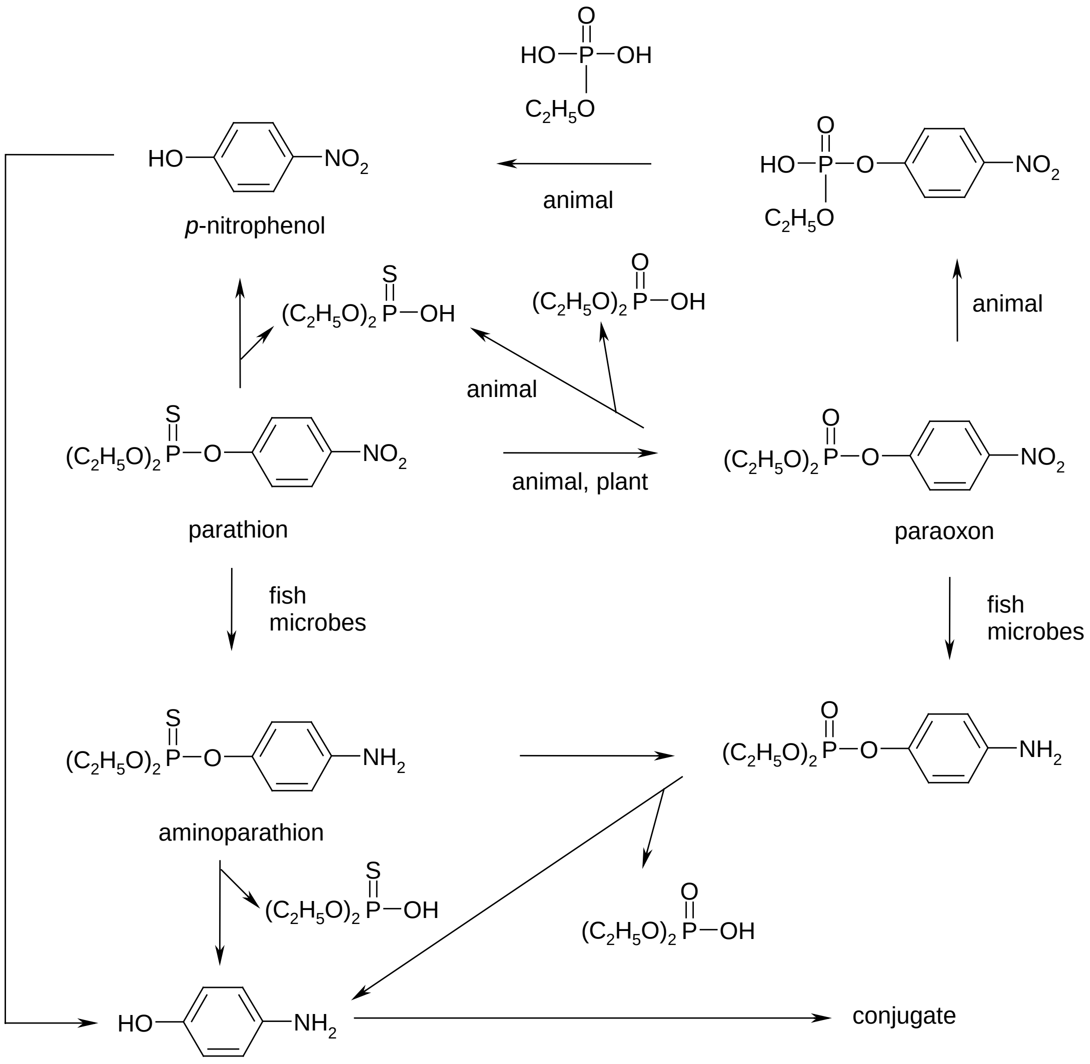
Table 6-3. Bioconcentration Data for Parathion

Species common name/ scientific name	Exposure type	Exposure concentration (µg/L)	Duration (days)	BCF	Reference
Freshwater fish					
Bluegill/*Lepomis macrochirus*	F	510	0.5	63	HSDB 2013
Bluegill/*L. macrochirus*	F	640	3	462	HSDB 2013
Brook trout/*Salvelinus fontinalis*	F	3,180	0.33	68	HSDB 2013
Brook trout/*S. fontinalis*	F	270	5.83	344	HSDB 2013
Killifish/*Oryzias latipes*	F	–	1–3	98[a]	Tsuda et al. 1995
Tadpoles	I	64			Hall and Kolbe 1980

[a] Average BCF value reported in the cited reference.

BCF = bioconcentration factor; F = flow-through exposure system; I = immersion

Figure 6-2. Degradation Pathways for Parathion



Source: EPA (1977b)

The mobility of parathion in soils and sediments is expected to be low based on measured K_{oc} values. Parathion was reported to have average K_{oc} values of 674 and 1538 in Israeli soils (0.11–5.82% organic matter) and sediments (3.08–7.85 organic matter), respectively (Gerstl and Mingelgrin 1984). In four soils with an organic carbon range of 0.087–0.65%, parathion had an average K_{oc} of 10,454 ppm (Hamaker and Thompson 1972). Mingelgrin and Gerstl (1983) reported a K_{oc} range of 314–15,860 for parathion for an unspecified number of soils with an organic content ranging from 0.2 to 6.1%. In four soil types with organic carbon content ranging from 0.41 to 43.7%, parathion had K_{oc} values ranging from 965 to 1,700 (Sharom et al. 1980). In five sterilized Iowa soils (organic matter content 0.88–31.65%), parathion had K_{oc} values ranging from 602 to 805 (Felsot and Dahm 1979). Chu and Chan (2000) reported a K_{oc} value of 10,700 mL/g for parathion. A log K_{oc} of 3.20 (K_{oc} of 1,585) was reported by Sabljic et al. (1995). Adsorption patterns indicated that parathion has an initial fast adsorption reaction occurring within 4 hours. Adsorption was almost complete 1 hour after application (~86% of added parathion) and equilibrium (88% of added parathion) was reached within 4 hours (Saffih-Hdadi et al. 2003).

Because this insecticide is highly adsorbed by soils, leaching into groundwater is expected to be minimal. Additional parameters influencing the leaching potential of this chemical include the soil type (e.g., clay versus sand), amount of rainfall, depth of the groundwater, and the extent of degradation (Kenaga 1980). Sorption of parathion is positively correlated with organic matter content of the soil (Felsot and Dahm 1979). Parathion had a mobility of 0.01 compared to that of water in a French soil (Moreale and Van Bladel 1983) and ranked 36 and 40 in a ranking of 41 pesticides by attenuation factor and retardation factor, respectively, in two sandy soils (Rao et al. 1985). The fraction of parathion leached from soil by 10 successive 200-mL applications of water to a soil column was 1.24 and 4.36% for an organic soil and sand, respectively (Sharom et al. 1980). Only a small fraction (10%) of parathion adsorbed to a sterile sandy loam was found to undergo diffusion (the diffusion constant in soil with highest moisture content was 0.03 cm^2/day) (Gerstl et al. 1979). In soil columns of Nacogdoches clay subsoil, parathion leached to 60 inches when 230 inches of rainfall was simulated, while in Houston Black clay surface soil, 1,725 inches of rain were required to produce leaching to 60 inches. Parathion movement in those soils under saturated conditions was slow, with parathion dissolving slightly in the liquid phase while its downward movement was retarded primarily by adsorption. The authors considered that, under normal unsaturated flow field conditions, the downward flow of parathion in water would be offset by its upward movement in water vapor (Swoboda and Thomas 1968). Parathion applied at 0.1 lbs/acre, followed by flooding and a subsequent application of 0.2 lbs/acre, was degraded before reaching drainage tiles 6 feet below the soil surface (EPA 1977b). In field studies, little leaching occurred in a sandy loam soil in 16 years after four annual applications of parathion despite 42 inches of precipitation per year (Stewart et al. 1971). Little parathion was found below 9 inches, 6 years after 30,000–95,000 ppm was applied to the soil (Wolfe et al. 1973). In a 15-year study of residues in a light sand soil, no parathion was found to have leached below 8 inches (EPA 1977b). In an 8-month persistence test under experimental conditions with 20 cm of simulated rain, no parathion was found below 1 inch. In cases where small amounts of parathion penetrated into the soil, it was believed to be the result of the movement of particulate or microparticulate matter containing sorbed parathion rather than by leaching (EPA 1977b). In a field study involving the application of parathion to a peach orchard providing watershed for a 2.7-acre pond, no residue was found below 6 inches and there appeared to be little desorption of the insecticide from the bottom sediment of the pond (Faust and Suffet 1966).

6.3.2 Transformation and Degradation

Parathion is subject to a variety of abiotic and biotic degradation processes in all environmental compartments. The adsorption of parathion to suspended solids and sediments in the water column may affect its rate of volatilization, photolysis, biodegradation, and its bioavailability to aquatic organisms (Schuurmann et al. 2006). It follows two major fate pathways: degradation to less toxic compounds or oxidative conversion to the toxic bioactive product, paraoxon (CDFA 1988). The extent of conversion to paraoxon is dependent upon the amount of sunlight, atmospheric oxidants present, and type of formulation applied (CDFA 1988).

6.3.2.1 Air

Parathion in the atmosphere may be subject to direct photolysis since it absorbs light in the spectra above 290 nm. Under atmospheric conditions, oxidation especially influences the transformation rates of parathion to its degradation product, paraoxon (Mansour et al. 1983). Parathion conversion to paraoxon occurs rapidly in air, is promoted by sunlight, and takes place largely in the vapor phase (Seiber and Woodrow 1984). The presence of ozone catalyzes the conversion of parathion to paraoxon. While at normal ozone levels (30 ppb), paraoxon production was quite low (approximately 2.1–4.1% in 8 hours), at ozone levels found under smog conditions (300 ppb), 10–65% conversion was found in 8 hours (HSDB 2013). The photolysis half-life of parathion as determined in a laboratory photoreactor was 41 minutes. This half-life was reduced to 23 minutes in the presence of >1 ppm ozone (Woodrow et al. 1978, 1983). Field experiments performed by releasing parathion as an emulsifiable concentrate and sampling downwind resulted in a half-life of 5 minutes at 4 PM (early June), and 131 minutes after sunset

(Woodrow et al. 1978, 1983). The half-life (first-order kinetics) for the vapor-phase reaction of parathion with hydroxyl radicals in the atmosphere is estimated to be 4.2 hours, assuming an atmosphere containing 5×10^5 hydroxyl radicals/m^3 at 25°C (Meylan and Howard 1993).

6.3.2.2 Water

Parathion released to water may be subject to both abiotic degradation (i.e., hydrolysis and photolysis) and biotic degradation by microorganisms. Microbial metabolism is the major means of parathion detoxification in soil and aquatic environments (Sethunathan et al. 1977).

Parathion slowly undergoes hydrolysis to form toxic products, paraoxon and *p*-nitrophenol, which are much more soluble than the parent compound. At pH 7 and 20°C, the hydrolysis half-life of parathion was reported as 130 days (EPA 2002). A hydrolysis half-life of 62 days was reported for parathion in dilute solutions at pH 6.3 and 20°C. It was also reported to hydrolyze at 1% in river water at pH 7.2, with a half-life of 60 days (Mansour et al. 1999). In a natural aqueous medium, approximately 70% of parathion was hydrolyzed in 4 weeks and 72% was hydrolyzed in 6 weeks (EPA 1977b).

The fate of parathion in water is dependent upon physical factors such as temperature, pH, and microbial composition. Parathion is relatively stable in neutral or acidic pH range, but is hydrolyzed rapidly in alkaline conditions (Sethunathan et al. 1977). Chapman and Cole (1982) reported the half-lives of parathion in sterile aqueous buffer solutions to be 39, 43, 33, 24, and 15 weeks at pH levels of 4.5, 5, 6, 7, and 8, respectively. Fisher and Lohner (1987) studied factors affecting the stability of parathion in the aquatic environment, with emphasis on pH. In water of aquatic microcosms adjusted to pH 4, 6, and 8, pH was found to be insignificant in controlling levels of parathion. After 7 days, parathion accounted for 29.7, 28.7, and 36.6% of total radioactivity in the water of microcosms held at pH 4, 6, and 8, respectively. In abiotic water, however, no parathion breakdown was observed in 40 days at any of the pH levels, demonstrating the importance of biotic factors, particularly microorganisms, in degrading parathion (Fisher and Lohner 1987). A half-life of 8.77 days was reported for parathion in an experiment using samples of non-filtered Limon River water that was exposed to sunlight and in contact with the atmosphere (Medina et al. 1999).

Temperature has been shown to be an important factor in the degradation of parathion through elevation of the rate of hydrolysis. An increase in temperature from 20 to 37.5°C (pH 7.4) resulted in a decrease in the half-life of parathion from 130 to 27 days in an aqueous system (Freed et al. 1979).

The presence of metal ions can have a catalytic effect on hydrolysis as was shown by the decrease in half-life of formulation parathion sprays in the presence of copper ions and in other experiments done in the presence of copper and calcium ions (Plastourgou and Hoffmann 1984). However, it was not clear how much of an effect the presence of metal ions will have in natural waters. The half-life for chemical hydrolysis in sterile seawater was reported to be approximately 1 year at 4°C and pH between 7.8 and 8.8, showing that the pH was not a demonstrable factor in degradation (Wade 1979). It was reported, however, that divalent cations may have catalyzed the hydrolysis.

The rate of aqueous photolysis is dependent upon the intensity and wavelength distribution of sunlight, which varies by the time of day and season of the year. The presence of natural photosensitizers, such as humic and fulvic acid also affects the potential rate of photolysis of parathion in natural waters. Measured photodegradation half-lives for parathion in lake water, river water, marine water, groundwater, and distilled water were 17.8, 23.7, 18.9, 21.6, and 19.6 days, respectively. Paraoxon was found to be the primary degradation product by this process, while small amounts of *p*-nitrophenol, aminoparathion, and triethyl phosphothioate were also observed (Sakellarides et al. 2003). Half-lives of 120 and 86 days were observed when parathion was incubated in pH 7.3 river water in darkness at 6 and 22°C, respectively; the half-life decreased to 8 days when parathion was incubated in sunlight in pH 7.3 river water (Lartiges and Garrigues 1995). Half-lives of 542 and 44 days were observed when parathion was incubated in pH 8.1 seawater in darkness at 6 and 22°C, respectively; a half-life of 18 days was observed when parathion was incubated in seawater in sunlight (Lartiges and Garrigues 1995). Irradiation for 10 hours in aerated distilled water resulted in 88% degradation attributed to photolysis (Mansour et al. 1983).

Photosensitizers that are present in eutrophic natural waters accelerate photolysis. In river water, parathion had a photolysis half-life of 1.2 days, and this half-life was reduced to 0.69 days in the presence of the photosensitizer, riboflavin (Zhao and Hwang 2009). While 20% of parathion in distilled water was lost by photolysis in 18 hours, the same loss occurred in only 2 hours in Okeefenokee Swamp water (Zepp and Baughman 1978). The presence of hydrogen peroxide at concentrations that occur naturally in agricultural irrigation water and other surface water has been shown to increase the rate of photodegradation. The addition of hydrogen peroxide to distilled water reduced parathion remaining in solution from 65 to 28% when exposed to October sunlight for 245 hours (Draper and Crosby 1984).

Parathion has been shown to biodegrade in acclimated natural waters within several weeks. Parathion (5 ppm) completely degraded within 2 weeks in acclimated water from Holland Marsh, a vegetable growing area in Ontario, being almost quantitatively converted to aminoparathion, while only 10% degradation occurred in 16 weeks when the water was sterilized (Sharom et al. 1980). In the waters of the Little Miami River, Ohio, a small stream that receives domestic and industrial waste as well as farm runoff, 50% of parathion (10 ppb) degraded in 1 week and none could be detected after 4 weeks (Eichelberger and Lichtenberg 1971).

In non-acclimated seawater, marine plankton were responsible for both chemical and biological degradation of parathion. Surface water of varying salinity (0–28 ppt) collected from the Mississippi Sound estuary system degraded parathion with a 45-day half-life at 30°C and was shown to be independent of salinity (HSDB 2013). The half-life of parathion in sterile seawater was reported as approximately 1 year, but was reduced to 56 days under nonsterile conditions (Wade 1979). After 30 days of incubation in non-sterile coastal river water (24 ppt salinity), only 21, 14, and 6% of parathion remained at pH values of 6, 7, and 8.16, respectively, while in sterile coastal river water, 64, 57, and 49% of the initial parathion remained at pH 6, 7, and 8.16, respectively (Wang and Hoffman 1991). In an experimental study performed by Carvalho et al. (1998), the persistence half-life of parathion dissolved in marine water at 32°C was 9–46 days, depending on the salinity of the water.

Parathion has been shown to degrade in activated sludge treatment plants. With adequate aeration, high levels of parathion wastes were destroyed within 7–10 days in a treatment plant (Sethunathan et al. 1977).

6.3.2.3 Sediment and Soil

Once released to soils and sediments, parathion can be degraded by hydrolysis, photolysis, and biodegradation by several genera of microorganisms. In an experiment studying the relative decomposition rates of ^{14}C-labeled parathion in sterilized and non-sterilized soil, degradation was principally of biological origin based on measurements of cumulative $^{14}CO_2$ released. At 7 days after application, <3% of abiotic degradation was observed for parathion in sterilized soil compared with 18% in non-sterilized soil. After 56 days of incubation, the respective values were relatively much closer (34% and 55%), indicating that there was less parathion available for biological activity and that the biodegradation pathway may be cometabolic. Parathion may be transformed by reduction to aminoparathion, which is then hydrolyzed to 4-aminophenol, or by oxidative desulfurization to paraoxon, which is hydrolyzed to *p*-nitrophenol, depending on the type of soil and, therefore, the type of microbial biomass (Saffih-Hdadi et al. 2003).

Moisture content of soil and initial concentration of parathion can have an effect on the rate of degradation and evaporation, both of which contribute to removing parathion. The elimination of parathion in soil with three different moisture contents increased with initial concentration (1.4–28 ppm) and moisture content (Gerstl et al. 1979). The percentage of parathion eliminated after 11 days of incubation ranged from 96% at high initial concentrations and moisture content to 20% at low concentrations and moisture levels.

Based on several degradation studies using sterile and nonsterile samples and isolated cultures of microorganisms, it has been shown that microbial metabolism is the major means of parathion detoxification in soil and aquatic environments (Sethunathan et al. 1977). After 8 weeks of incubation in an organic and a mineral soil, <2 and 6% of the 1 ppm parathion applied remained, respectively, while 80 and 95% remained in sterilized controls, respectively (Chapman et al. 1981). Miles et al. (1979) reported the half-lives of parathion (10 ppm) in a sandy loam and organic soil to be <1 and 1.5 weeks, respectively, with only 5% remaining after 3 and 10 weeks, respectively. During three composting trials in 1979, 1982, and 1983 consisting of 56–75-day composting periods, composting of organic wastes was characterized by very high biological activity (Vogtmann et al. 1983).

Parathion has also been shown to degrade under anaerobic conditions with reduction of parathion to aminoparathion as the main degradation pathway. When parathion (500 ppm) was incubated in flooded anaerobic alluvial soil, 43 and 0.09% remained after 6 and 12 days, respectively, with degradation occurring by reduction to aminoparathion (Adhya et al. 1981a). In a parallel experiments in which parathion was incubated for 30 minutes in soil suspensions of five 30-day flooded (anaerobic) soils and aerobic soils that had been previously reduced, no degradation occurred in the aerobic soils, while 35–68% degradation occurred in the anaerobic soils (Adhya et al. 1981b). The most reduced soils produced the most rapid degradation. The effect of sulfur content of the soils was also investigated in the study. In anoxic sulfur-containing environments such as flooded acid sulfate soils, hydrogen sulfide evolved as the end product of anaerobic metabolism of sulfate, readily reacted with the aminoparathion degradation product to form desethyl aminoparathion (Adhya et al. 1981b). After repeated application of parathion to flooded soils, the degradation pathway shifted from reduction to hydrolysis (Adhya et al. 1981c).

Two microorganisms isolated from flood soils also were found to hydrolyze parathion (Adhya et al. 1981c). Parathion was rapidly hydrolyzed within 24 hours by both *Flavobacterium* sp. and *Pseudomonas* sp. A hydrolysis product of parathion, *p*-nitrophenol, was not metabolized further by the *Flavobacterium* sp., while the *Pseudomonas* sp. readily metabolized *p*-nitrophenol to yield nitrate (Adhya et al. 1981c).

Studies with sterile soils and clay minerals have shown that parathion may undergo slow chemical hydrolysis in soil systems. During 130 days of incubation in gamma irradiated soils (30 kGy) of differing organic matter and clay mineralogy at room temperature, 3–23% of parathion was hydrolyzed in air-dried soil and <10% was hydrolyzed in moist soils. Among the soil constituents, clay and organic matter (kaolinite > montmorillonite > organic matter) were the most important in catalyzing the chemical hydrolysis of parathion in sterile soils. However, adsorption follows the reverse order, indicating that catalysis occurs at active absorption sites at the soil surface (Sethunathan et al. 1977). In 14 soil studies, 3–33% of the initial parathion was degraded after 130 days of incubation in dry soil (Yaron 1975). For most of the soils, the presence of water hindered degradation, presumably by blocking active sites on the soil (Yaron 1975). Parathion degradation decreased as moisture content of various kaolinite clays increased until a moisture content equivalent to the upper limit of bound water was reached (Saltzman and Mingelgrin 1984). As moisture content was increased beyond this point (e.g., in flooded conditions) the hydrolysis rate of the parathion sharply increased (Saltzman and Mingelgrin 1984).

Although the treatment of crops with pesticides such as parathion often resulted in significant contamination of the adjacent soils, photolysis is only an important environmental fate process for contaminated surface soils since sunlight is rapidly attenuated and does not penetrate much beyond the soil surface. Measured photodegradation half-lives for parathion in sandy clay loam (0.90% organic matter), clay loam (1.94% organic matter), and sandy loam (3.52% organic matter) were 21.3, 15.6, and 20.8 days, respectively (Sakellarides et al. 2003). No correlation was observed between the percentage of organic matter present in the soil and the rate of photodegradation.

6.3.2.4 Other Media

Thin films of parathion that may be formed on leaves and other surfaces after spraying have a photodegradation half-life of 88 hours (Chen et al. 1984). Conversion products on leaf surfaces and dry dust particles in field tests are paraoxon and *p*-nitrophenol (Crosby 1979). In a photodegradation study, parathion applied to epicuticular leaf and fruit wax of a tomato degraded 16.3 and 20.7% after 8 hours of exposure, respectively. The presence of the wax interacts with parathion at the nitro group that participated in the photoreduction, leading to the formation of the azo derivative 4,4’-bis(di-ethoxy-phosphinothioyloxy)azobenzene (Fukushima and Katagi 2006).

6.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to parathion, which has not been used in the United States for several years, depends in part on the limits of detection of supporting analytical data and appropriate sampling of environmental media and biological specimens. Concentrations of parathion 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 parathion 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. The analytical methods available for monitoring parathion in a variety of environmental media are detailed in Chapter 7.

Care should be taken when assessing analytical results for which a limit of detection (LOD) or similar sensitivity value is not provided for the substance of interest, and the study reports not having detected that substance; failing to detect a substance does not mean that it is not present.

Most information on parathion concentrations in various environmental media derived from large-scale US monitoring networks dates from before the mid-1990s and likely no longer reflects current conditions. There is a noticeable lack of national monitoring studies that would allow meaningful estimation of current parathion concentrations associated with various environmental media. Reliable evaluation of the potential for human exposure to parathion depends, in part, on the LOD of supporting analytical data and appropriate sampling of environmental media and biological specimens.

6.4.1 Air

When parathion was still being used as a registered pesticide in the United States, a range of ambient air concentrations of 0.017–0.089 μg/m^3 was reported (CDFA 1988). Ambient air concentrations of parathion and its oxidation product, paraoxon, converted to parathion were measured by the Air Resources Board in the San Joaquin Valley, California from January 6 to February 14, 1986 and in the Imperial Valley, California from September 23 to October 22, 1986. In this study, the highest individual 24-hour values measured in the northern San Joaquin Valley for parathion and for combined parathion plus converted paraoxon were 0.834 and 1.423 μg/m^3, respectively, with means of 0.141 and 0.170 μg/$m^3$for all samples collected from the six sites sampled over the 23-day study period. In the southern San Joaquin Valley and in the Imperial Valley, respectively, the peak amounts of parathion (excluding converted paraoxon) in the air were considerably lower; the highest measured individual 24-hour values were 0.089 and 0.150 μg/m^3 with means of 0.023 and 0.025 μg/m^3 for each area. The low values in the southern San Joaquin Valley were potentially the result of the wet, foggy weather prevailing during the winter months. In the Imperial Valley, the low ambient levels of parathion could have been the result of increased volatility during the very warm weather of the fall months (CDFA 1988). Ambient parathion concentrations were measured under foggy atmospheric conditions in and around the Central Valley of California (Parlier, California), which is a prime agricultural area dominated by fruit, nut, and citrus orchards (Glotfelty et al. 1990; Seiber et al. 1993; Zabik and Seiber 1993). Parathion was detected in air samples collected from Parlier, California on January 12, 1986 under foggy atmospheric conditions at a total concentration of 9.4 ng/m^3, with 78.5% existing in the vapor phase (Glotfelty et al. 1990). Zabik and Seiber (1993) studied the atmospheric transport of parathion from California's Central Valley to the Sierra Nevada Mountains. Air samples collected from January through February 1991 represented the simultaneous collection of both vapor and particulate phases. Concentrations of parathion and paraoxon were 26–13,000 pg/m^3 (0.026–13 ng/m^3) and 8–3,800 pg/m^3 (0.008–3.8 ng/m^3), respectively, for samples collected at the 114-m elevation and below the LOD and 71 pg/m^3 (<LOD–0.071 ng/m^3) and <LOD–220 pg/m^3 (<LOD–0.22 ng/m^3), respectively, at the 533-m elevation. The pesticide concentrations in air samples decreased with distance and elevation moving east from the Central Valley into the higher elevations of the Sierra Nevada Mountains. At times, air concentrations of parathion at the 114-m elevation were 1,000 times greater than concentrations detected at the 533-m elevation. Concentrations at the 1,920-m elevation were typically below the limit of quantification for parathion, but paraoxon was detected at concentrations of up to 10 pg/m^3. The higher paraoxon than parathion concentrations at higher elevations indicate that the oxygenation process occurs during atmospheric transport resulting in increasing conversion of parathion to paraoxon as residence time in the atmosphere increases. Wet deposition samples (rain and snow) collected at the 114-m elevation contained up to 7,600 pg/mL parathion and 8,300 pg/mL paraoxon. Seiber et al. (1993) reported an average parathion concentration of 63.5 ng/m^3 in 24-hour ambient air samples collected near dormant orchards in the northern San Joaquin Valley, California during 17 days in January 1989, the most intensive period of dormant orchard spraying. The average day- and night-time concentrations were 52.0 and 119.6 ng/m^3, respectively. Those values are significantly lower that reported by the California Department of Food and Agriculture (CDFA) for the same location and time frame (CDFA 1988).

In a study of rain and air samples collected from agricultural and urban sites in Mississippi during April to September 1995, parathion was not detected (method reporting level of 24 pg/m^3) in urban air or rain and agricultural air. It was detected once in agricultural rain samples (Coupe et al. 2000; Foreman et al. 2000).

Parathion was included in the National Air Pesticide Monitoring Program from 1970 to 1972 at selected sites in 14–16 states. In 1970, 3.2% of 787 samples tested positive at a mean concentration of 64.2 ng/m^3 and a maximum concentration of 834 ng/m^3. In 1971, 2.3% of 667 samples tested positive at a mean concentration of 9.3 ng/m^3 and a maximum concentration 109 ng/m^3. In 1972, parathion was not detected in 1,025 samples (Kutz et al. 1976); however, the LOD was not provided. Stanley et al. (1971) reported the results of a pilot study of atmospheric contamination of pesticides, in which air was sampled at nine localities in the United States, representing both urban and agricultural areas. Parathion was detected in only one location (Orlando, Florida) at a maximum concentration of 465 ng/m^3 (37% samples positive). In a study of pesticide levels in ambient suburban air, parathion was detected in Miami, Florida (60% positive, 2.8 ng/m^3 mean, 12.1 ng/m^3 maximum), but not in Jackson, Michigan or Fort Collins, Colorado (Kutz et al. 1976); the LOD appears to be <0.4 mg/m^3 since that value, for Pennsylvania, was the lowest positive result reported in the study.

In addition to its presence in the ambient atmosphere, parathion has also been monitored in both outdoor and indoor air associated with its use in occupational exposure situations. The highest levels of parathion measured when it was still registered for use in the United States were reported in the wash down area for crop-dusting aircraft (320 μg/m^3) and in the cockpits of those aircraft during application (179 μg/m^3). These levels were followed by air levels of 48 and 43 μg/m^3 measured in the truck cab and tractor towing spray-rigs, respectively (CDFA 1988). Lewis and Lee (1976) reported concentrations of parathion in a formulating plant and storage shed in South Florida to be 557 and 48.9 ng/m^3, respectively. Mean air concentrations in open and closed tractors with an oscillating boom during spray application of parathion on citrus trees ranged from 4 to 93 μg/m^3 (Carman et al. 1982).

Reported levels of parathion in the air at a site of application ranged from 2 to 18 μg/m^3 within a day of application and decreased to a range of not detected (LOD not stated) to 0.005 μg/m^3 21 days after application. Reported off-site downwind levels ranged from 34 μg/m^3, 40 yards from the sprayed field during application, to 0.002 μg/m^3 100 yards from the field 6 days after application (CDFA 1988). The highest parathion concentration found by one study within an orchard during spraying and dusting operations was 0.74 mg/m^3. Other studies found parathion concentrations as high as 15 mg/m^3 during spraying and dusting (Wolfe 1976). Concentrations of parathion in the air of a plum orchard were 3,500 ng/m^3 immediately after spraying and 4,100, 394, 149, 21, and 16 ng/m^3 1, 2, 5, 14, and 21 days after spraying, respectively (Seiber and Woodrow 1984). Downwind 100 m from the plum orchard, parathion concentrations of 35, 9, 1.6, and 0.9 ng/m^3 were detected 2, 3, 6, and 21 days after spraying, respectively.

The Colorado Community Pesticide Sampling Programs detected parathion in monthly air samples taken inside and outside the homes of 12 men who were occupationally exposed to parathion either as formulators or farmers. The higher levels were found for formulators, with indoor exceeding outdoor levels for both groups (Tessari and Spencer 1971).

In the Canadian Atmospheric Network for Currently Used Pesticides, a nationwide air surveillance study of pesticides in agricultural regions in Canada, the highest concentrations of parathion were found at Vineland where the peak concentrations were 784 pg/m^3 in 2004 and 81.5 pg/m^3 in 2005 (Yao et al. 2008). Parathion may be found in environmental media in agricultural regions of other countries that still use parathion, indicating that it could be brought into the United States (e.g., on produce).

6.4.2 Water

Spray drift, runoff, and erosion flux may result in the contamination of streams and adjacent water bodies in locations where pesticides are applied. Leaching from the soil surface may also contaminate groundwater; however, streams and other surface waters appear to be more vulnerable to contamination than groundwater in most hydrologic settings (USGS 2007). During Iowa’s Ambient Surface Water Monitoring program that included about 80 sampling sites between 1999 and 2001, only one detection of parathion occurred and the concentration was low (in the 0.05 ppb range) (EPA 2002). In a USGS program for monitoring pesticides in the streams of the western United States for the period of October 1968 to September 1971 in which parathion was tested for quarterly at 20 stations, it was detected at 40 ppt in two samples from the Gila River, Arizona and at 40 and 160 ppt in two samples from the Sacramento River, Verna, California (Schulze et al. 1973). Parathion was not detected in any water samples collected from July to September 1984 from Little Miami River, Ohio above or below a municipal waste water outfall, although the LOD was not reported (HSDB 2013). No parathion was detected in water or particulate matter samples collected from Lake Superior or Lake Huron at quantification limits of 5 ppt and 100 pg, respectively (Glooschenko et al. 1976). In the Erie River Basin, parathion was not detected (LOD not reported) in the over 100 samples collected before 1974 (Waldron 1974). Parathion (LOD not reported) was detected in 1 of 174 sampling stations across the nation’s rivers collected prior to 1985 during a USGS water supply study (HSDB 2013). In a monitoring program by the California Department of Water Resources, the highest concentration of parathion detected in Sacramento-San Joaquin Delta Water was 0.035 μg/L (Lam et al. 1994). Parathion (LOD not reported) was not detected in any surface water samples collected during 1999–2000 as part of the USGS National

Water-Quality Assessment (NAWQA) Program in which samples were collected from 34 sites located throughout the Yakima River Basin, Washington (Ebbert and Embrey 2001). Parathion was detected in 1 of 215 surface water samples (Cherry Creek, Denver, Colorado) at a concentration of 0.014 μg/L during a study monitoring eight urban streams in the United States from 1993 to 1994 (Hoffman et al. 2000).

Parathion was detected in ditch water draining cranberry bogs that had been treated with parathion in the Lower Fraser Valley of British Colombia, Canada, at concentrations ranging from not detected to 21 μg/L (Wan et al. 1995). In farm ditch water collected from seven locations in the Lower Fraser Valley, British Columbia in 1991, parathion was detected at mean concentrations of 0.13 and 0.20 μg/L at Westham Island and Colverdale, respectively (Wan et al. 1994).

Water samples collected from 15 lakes and 2 creeks on the Sparta, Illinois National Guard Armory, which is surrounded primarily by agricultural land, in the winter of 2003 contained a mean parathion concentration of 17 ng/L (4 of 39 samples). Parathion (LOD not reported) was not detected in 42, 41, and 33 samples collected in the spring, summer, and fall 2003, respectively (Ownby et al. 2004). Parathion was not detected in water samples collected from tributaries in Sothern and Central Ontario (LOD = 4.6–15.5 ng/L) during sampling conducted from 2003 to 2008 by Environment Canada (de Solla 2012a, 2012b).

Parathion has been included in several state monitoring programs. In a groundwater quality study conducted in a rural region of Baltimore County by the Maryland Geological Survey, parathion was detected above the minimum reporting limit (0.004 μg/L) in 1 out of 112 samples, at a concentration of 0.022 μg/L (EPA 2002). Parathion (LOC not reported) was not detected in ground water from five shallow monitoring wells, where the insecticide was used within 300 feet of the wells, sampled in North Carolina between 1991 and 1995 (EPA 2002). In a groundwater monitoring program run by the North Dakota Department of Health that collected approximately 2,700 samples from 1,465 wells between 1992 and 2001, parathion was detected in 3 samples from 2 wells at concentrations of 0.274, 0.322, and 1.833 μg/L (EPA 2002).

No parathion was detected (LOD 0.5 ppb) in 36 drinking water wells sampled in Hawaii in March, 2001 (EPA 2002). In Iowa’s Statewide Rural Well-Water Study (LOD not reported) that included 686 private wells sampled once during 1988–1989, with 10% of the wells repeat-sampled during 1990 and 1991, no parathion was detected (EPA 2002). No parathion was detected in drinking water (LOD not reported) in the FDA Total Diet Study for infants and toddlers conducted between 1980 and 1982 and the FDA Total

Diet Study for adult groups conducted from 1980 to 1982 (Gartrell et al. 1986a,1986b). No parathion was detected in 54 monitored wells (primarily municipal) (LOD not reported) in selected California communities in the early 1980s (Maddy et al. 1982). A parathion concentration of 4.6 ppb was measured in a California drinking water well sample in 1979 (Burmaster 1982). Parathion was not detected in Ottawa tap water (LOD <1 ppt) in the late 1970s (Lebel et al. 1979).

Parathion concentrations were measured under foggy atmospheric conditions in and around the Central Valley of California, which is a prime agricultural area dominated by fruit, nut, and citrus orchards Parathion was detected in atmospheric fog water collected from agricultural areas Parlier, Corcoran, and Lodi, California in 1985 at concentrations of 9,000, 950, and 184,000 ng/L, respectively (Glotfelty et al. 1987). It was also detected in fog samples collected near Parlier, California in January 1986 at concentrations of 3.6, 31, 30, 2.7, 39, and 23 μg/L (Glotfelty et al. 1990). Seiber et al. (1993) reported parathion concentrations ranging from 4.3 to 19.0 ppb in fog water samples collected from the San Joaquin Valley, California in January 1989 during the spraying of near dormant orchards.

6.4.3 Sediment and Soil

Parathion has not been the focus of many national soil or sediment monitoring programs in the United State, but has been monitored in regional studies associated with agricultural applications in both the United States and Canada. Parathion has also been reported in the growing soil of crops such as chili peppers and tomatoes imported from other countries (e.g., China, India, and Mexico) into the United States (Diggory et al. 1977; Liu et al. 2016; Mamta et al. 2015a, 2015b; Reiler et al. 2015; Steiniger et al. 2010; Velasco et al. 2014). In the 1972 National Soils Monitoring Program, which included 1,246 samples of cropland soil in 37 states, parathion was detected in 0.6% of the samples at mean and maximum concentrations of <0.01 and 19 ppm, respectively (Carey et al. 1979a). Parathion (LOD not reported) was not detected in the 1971 Urban Soils Monitoring Program that sampled soils from five U.S. cities (Carey et al. 1979b). In a 1972 survey of rice growing soils (99 samples) in five states, parathion was detected in Arkansas (4.2% of samples) and California (10% of samples) at mean concentrations of 0.01 and <0.01 ppm (dry weight), respectively, and maximum concentrations of 12 and 0.01 ppm, respectively (Carey et al. 1980). Velasco et al. (2014) evaluated organochlorine pesticides in agricultural soil in Mexico and reported finding parathion in 62% of samples with an average concentration of 47 μg/kg.

Parathion was detected in the organic soil of 12 of 28 farms in six vegetable growing regions of southwestern Ontario in 1976 at concentrations ranging from 6 to 2,500 ppb (Miles and Harris 1978). In Southern Ontario orchard soil, parathion was detected in 5% of the apple orchards sampled at a mean concentration of 5 ppb (Carey et al. 1980). Frank et al. (1976) reported that trace amounts of parathion residues (<0.001 ppm mean concentration, 0.021 ppm maximum concentration) were found in the upper 15 cm of soils from 31 apple and 16 sweet cherry orchards in Southern Ontario collected between 1972 and 1975. No parathion was detected in subsurface (15–30 cm) soils (LOD not reported). In soil samples collected from an evaporation pit in California between May and September 1985, parathion was detected at concentrations of 1,064–1,972 ppm at 0–7.5 cm surface soils, 51–60 ppm at 7.5–15 cm below the soil surface, 37 ppm at 22.5–30 cm below the surface, and 18 ppm at 60–67.5 cm below the soil surface (Winterlin et al. 1989). Parathion was detected in crop soils collected from the Lower Fraser Valley, British Columbia, Canada, between July and December 1991 at mean concentrations of 10, 19, 15, and 1,419 μg/kg in Westham Island, Ladner, Burnaby, and Cloverdale, respectively (Wan et al. 1994). Krapac et al. (1995) reported that parathion was detected in 4 of 822 soil samples collected from 49 agrichemical facilities located throughout Illinois at concentrations ranging from 69 to 5,540 μg/kg and a median value of 805 μg/kg.

Parathion was under the LOD of 0.1 ppb at industrial and agricultural locations in 11 sediment samples taken from the Delaware River estuary during a 1980–1981 USGS study (Hochreiter 1982). It was also not detected in any sediment samples collected from Lake Superior or Lake Huron during the summer of 1974 with a quantitation limit of 20 ppb (Glooschenko et al. 1976). Parathion was detected in bed sediments and suspended sediments collected from three locations in the Windrush River catchment, Oxforshire, United Kingdom in 1992 at concentrations of 0.3, 0.6, and 1.0 μg/kg and 13, 3.3, and 8.8 μg/kg, respectively (House et al. 1992).

Parathion (LOD not reported) was not detected in sediment in irrigation water collected from lagoons sampled from corn and sorghum fields in Kansas in 1974 (Kadoum and Mock 1978). Parathion was detected in sediments in a ditch draining cranberry bogs that had been treated with parathion in the Lower Fraser Valley of British Colombia, Canada, at concentrations ranging from not detected to 515 μg/kg (Wan et al. 1995). In farm ditch sediments collected from seven locations in the Lower Fraser Valley, British Columbia in 1991, parathion was detected at a concentration of 8 μg/kg (Wan et al. 1994).

6.4.4 Other Environmental Media

Parathion (LOD not reported) was not detected in fish fillets collected from 15 lakes and 2 creeks on the Sparta, Illinois National Guard Armory, which is surrounded primarily by agricultural land, in 2003 (Ownby et al. 2004). IARC (1983) reported that fish collected from 144 estuaries throughout the United States contained mean parathion levels of 10–75 ppb. Fish and shellfish specimens collected in May 1973 in the United States contained 10–40 ppb (IARC 1983). Clam, rainbow trout, and oyster samples collected from Massachusetts contained parathion concentrations of 20–30, 15–17, and 38–41 ppb, respectively (IARC 1983). In a nationwide estuarine fish monitoring program conducted from 1972 to 1976, 1,524 samples from 144 primary and secondary estuaries in 19 coastal states, parathion was detected in 1 of 39 samples from Connecticut, 1 of 251 samples from North Carolina, and 3 of 51 samples from Texas at mean concentrations of 10, 12, and 75 ppb, respectively (Butler and Schutzmann 1978). Parathion was detected at a concentration of 10 ng/g in oysters from Waikane Stream in Kaneohe Bay, Hawaii (Hunter et al. 1995). Parathion was detected in 2 of 20 samples of rodent muscle (40–110 ppb), 3 of 19 whole lizards (10–100 ppb), and 6 of 19 samples of bird muscle (10–210 ppb) obtained from an unspecified location within the state of Texas (HSDB 2013).

In the EPA's Revised Organophosphate Pesticides Cumulative Risk Assessment, a summary of residue monitoring data on organophosphate pesticides in food for the years 1994–2000 was reported (EPA 2002). The detection of parathion in these various foods and its concentration are presented in Table 6-4. The report also included a summary of FDA Total Diet Study analyses on organophosphate pesticides on meats for the years 1991–1999. No parathion residue (LOD not reported) was found in any of the tested meats (EPA 2002). Parathion was detected in 53 of 234 ready-to-eat foods tested repetitively for 10 years, 1982–1991, through the U.S. FDA's Revised Market Basket Survey at an average concentration of 0.0043 μg/g (HSDB 2013). In a survey of U.S. produce conducted from 1989 to 1991, parathion was detected in 13 of 6,970 produce samples, including apples (1 of 335), grapefruit (1 of 106), lemons (2 of 139), limes (1 of 78), oranges (6 of 220), peaches (1 of 84), and strawberries (1 of 76) (Schattenberg and Hsu 1992). It was also detected in 39 (2%) Total Diet Study foods between 1984 and 1986 (Gunderson 1995).

In the U.S. FDA pesticide residue monitoring study conducted from October 1993 to September 1994, which analyzed 11,348 domestic and import food samples from commodity groups, parathion was detected in unspecified foods (FDA 1995). In 1993–1994, the U.S. FDA conducted a study of pesticide residues in domestic and imported fresh apples and processed rice (LODs not reported). For apples,

Table 6-4. Parathion Residues in Various Foods from 1994 to 2000

Food item	Number analyzed	Number of detections	Average concentration (ppm)[a]	Maximum concentration (ppm)
Apple juice	1,344	0	0	0
Apples	2,263	5	8.1×10^{-5}	0.14
Apples (single serving)	377	0	0	0
Bananas	1,036	0	0	0
Broccoli	635	0	0	0
Cantaloupe	1,100	1	5.0×10^{-6}	0.005
Carrots	1,823	30	1.59×10^{-4}	0.044
Celery	143	0	0	0
Cherries	275	0	0	0
Corn syrup	430	0	0	0
Cucumbers	1,288	0	0	0
Grape juice	1,114	2	1.1×10^{-5}	0.007
Grapes	2,487	16	7.4×10^{-5}	0.043
Green beans (canned)	730	0	0	0
Green beans (fresh)	1,758	1	2.0×10^{-6}	0.003
Green beans (frozen)	639	0	0	0
Lettuce	1,580	1	1.2×10^{-5}	0.019
Milk	1,364	0	0	0
Nectarines	345	0	0	0
Oats (bran)	45	0	0	0
Oats (rolled)	287	0	0	0
Orange juice	1,212	0	0	0
Oranges	2,460	1	1.0×10^{-6}	0.003
Peaches (canned)	654	0	0	0
Peaches (fresh)	1,512	2	1.7×10^{-5}	0.022
Peaches (single serving)	534	1	2.1×10^{-5}	0.011
Peanut butter	716	0	0	0
Pears (canned)	647	0	0	0
Pears (fresh)	1,505	4	2.47×10^{-4}	0.31
Pears (single serving)	570	0	0	0
Pineapples	364	0	0	0
Potatoes	1,746	0	0	0
Poultry (adipose tissue)	476	0	0	0
Poultry (liver)	479	0	0	0
Poultry (muscle)	145	0	0	0
Rice	178	0	0	0

Table 6-4. Parathion Residues in Various Foods from 1994 to 2000

Food item	Number analyzed	Number of detections	Average concentration (ppm)[a]	Maximum concentration (ppm)
Soybean grain	748	0	0	0
Spinach (canned)	749	12	3.035×10^{-3}	1.6
Spinach (fresh)	1,385	0	0	0
Spinach (frozen)	715	1	2.4×10^{-5}	0.017
Strawberries (fresh)	1,768	0	0	0
Strawberries (frozen)	155	0	0	0
Sweet bell peppers	1,468	0	0	0
Sweet corn (canned)	652	0	0	0
Sweet corn (fresh)	19	0	0	0
Sweet corn (frozen)	635	0	0	0
Sweet peas (canned)	746	0	0	0
Sweet peas (fresh)	9	0	0	0
Sweet peas (frozen)	703	1	4.0×10^{-6}	0.003
Sweet potatoes	1,487	0	0	0
Tomatoes (canned)	737	0	0	0
Tomatoes (fresh)	1,766	5	2.9×10^{-5}	0.012
Wheat	1,563	1	1.4×10^{-5}	0.022
Winter squash (fresh)	1,078	5	3.2×10^{-5}	0.007
Winter squash (frozen)	343	2	4.1×10^{-5}	0.007

[a] Nondetects were counted as zero in calculating the average.

Source: EPA 2002

769 domestic and 1,062 imported samples were collected and for rice, 598 domestic and 612 imported samples were collected. Parathion was detected in one domestic apple sample at a concentration of 0.02 ppm and was not detected in any imported apple samples. Parathion was not detected in any rice samples (Roy et al. 1997). In an FDA survey of domestic and imported pears and tomatoes conducted between 1992 and 1993, parathion was detected in 1 out of 710 domestic and 2 out of 949 imported pears at maximum concentrations of 0.02 and 0.04 ppm, respectively, and in 2 out of 1,219 domestic tomatoes at a maximum concentration of 0.01 ppm, while it was not detected in 144 imported tomato samples (Roy et al. 1995). During a 5-year period from 1982 to 1986 in which the FDA Los Angeles District analyzed 19,851 samples of domestic and imported food and feed commodities, parathion was detected on 119 various agricultural commodities at concentrations ranging from 0.05 to 2 ppm (Luke et al. 1988). Parathion was not detected in 6,090 samples of fruits and vegetables tested in 2014 from the U.S. Department of Agriculture Pesticide Data Program (USDA PDP), which collects and tests domestic and imported foods for the presence of pesticide residues (USDA 2016).

In whole pasteurized milk collected monthly during 1990–1991 from 63 sampling stations located in large metropolitan areas throughout the United States, parathion was detected in 1 out of 2,739 samples at a concentration of 0.06 ppm (Yess et al. 1993).

6.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

Quantitative information from the TRI program for 2014 reports minor releases of parathion from hazardous waste treatment and disposal facilities in the United States (TRI14 2015). Inhalation exposure to parathion as a liquid or a dust at a hazardous waste treatment or disposal facility is unlikely due to its low vapor pressure, its degradation in the environment, and its high adsorption to soils. In order to mitigate the exposure and risk to the general population, especially children, the EPA terminated the production of most end use products as of December 31, 2002 (EPA 2000), and terminated the last registration for parathion products effective on December 31, 2006 (EPA 2006b). Because of these actions and environmental degradation processes, it is likely that neither the general population nor workers are exposed to parathion in the United States.

Parathion and its degradation products may be transported in the atmosphere and deposited to surface soils. Run off and erosion of soils containing parathion or other pesticides may contaminate nearby surface water bodies or leach into groundwater. Human exposure may result from contaminated surface or groundwater that is used in private wells or for a public water supply. Since parathion has not been able to be legally used in the United States since 2003, the likelihood of parathion concentrations exceeding a human health benchmark in a public supply well or domestic well is considered non-existent (EPA 2006b; USGS 2007).

The general public may have been exposed to parathion by dermal and inhalation exposure from spray drift onto adjacent area or by ingesting food containing parathion residues (HSDB 2013) during the time when parathion was used as a registered insecticide.

Given its cancellation of registered uses, the potential for human exposure to parathion through the diet and or drinking water is low (CDC 2009). During its use, intakes from diet and drinking water were reported to be below recommended limits. In 1982-84, a national U.S. FDA Total Diet Study was performed that showed the average daily intakes of parathion in the United States were 1.5, 1.4, and 2.0 ng/kg body weight/day, respectively, for females 14–16, 25–30, and 60–65 years of age and 1.2, 1.1, and 1.8 ng/kg body weight/day, respectively, for males 14–16, 25–30, and 60–65 years of age (Gunderson 1988). In the FDA Total Diet Study performed between July 1986 and April 1991, the average daily intakes of parathion in the United States were 0.0008, 0.0009, and 0.0016 μg/kg body weight/day, respectively, for males 14–16, 25–30, and 60–65 years of age and 0.0011, 0.0011, and 0.0018 μg/kg body weight/day, respectively, for females 14–16, 25–30, and 60–65 years of age (Gunderson 1995).). These studies used raw and commercially prepared foods, likely from both domestic and foreign sources where parathion was used, and purchased from retail suppliers in various regions of the country.

When it was in use, pesticide workers may have had much higher levels of parathion exposure following pesticide application compared to the general population. Exposure was often estimated by measuring the urinary metabolite *p*-nitrophenol; however, this substance is not unique to parathion as it is also a metabolite of methyl parathion and nitrobenzene. In a study of workers who handled parathion, end-of-shift urinary metabolite *p*-nitrophenol levels ranged from 190 to 410 μg/g of creatinine, a range of values approximately 2 orders of magnitude higher than levels found since 1999 in the U.S. general population (CDC 2015). Urinary levels of the metabolite *p*-nitrophenol ranged from <1 to 63 ppb (median <1 ppb; 41% detection in 974 samples) during the NHANES III study assessing exposure to a subset of general population adults from 1988 to 1994 (Needham et al. 2000). For survey years 1999–2000 and 2001–2002, no geometric mean urinary concentration of the *p*-nitrophenol could be calculated because the proportion of results below the detection limit was too high to provide a valid result (CDC 2009). The 95th percentile concentrations of *p*-nitrophenol in urine were 5.00 and 3.71 ppb in survey years 1999–2000 (sample size 1,989) and 2001–2002 (sample size 2,477), respectively. For survey years 2007–2008, the geometric mean urinary concentration of *p*-nitrophenol was 0.782 ppb for males (1,282 samples) and 0.582 ppb for females (1,282 samples) (CDC 2015). These levels decreased to 0.524 (1,342 samples) and 0.396 ppb (1,402 samples) for survey years 2009–2010 for males and females, respectively. Statistical results for survey years 1999–2000, 2001–2002, 2007–2008, and 2009–2010 are captured in Tables 6-5 and 6-6.

Workers historically were exposed to parathion primarily during field application and formulation of the insecticide before its use was banned. The dermal route is considered to be the most important route of entry during field applications, in formulation plants, and in other work situations where airborne pesticide drift is evolved, because of the disproportionately high levels of parathion which may drift onto exposed skin areas compared to the amount taken in during respiratory exposure. Dermal contact with treated surfaces such as leaves of sprayed crops is also an important route of exposure. Inhalation is the second most common route, especially when fine mists are formed as from concentrated spray (Wolfe 1976).

Workers of cotton fields were exposed to parathion at inhalation levels from 0.09 to 1.06 μg/30 minutes at 0–72 hours after application (Ware et al. 1973). Cab operators were exposed to average air concentrations of parathion ranging from 4 to 93 $\mu g/cm^3$ during the spraying of citrus crops (Carman et al. 1982). Mean dermal and respiratory exposure levels to parathion of various categories of workers were (category, dermal mg/hour, respiratory mg/hour): air blast operator, 18, 0.03; tractor driver hauling portable tower hand gun power sprayer during application, 12, 0.03; high pressure power hand gun spraying from tower position, 11, 0.03; high pressure power hand gun spraying from ground position, 47, 0.09; pilot dusting orchard, 13, 0.02; flagging for airplane spraying, 84, 0.02; operating tractor drawn boom ground duster, 8, 0.16; and operating tractor drawn boom ground sprayer, 4.7, <0.01 (Wolfe et al. 1967).

Dermal exposure through hand contact represented the greatest route of occupational exposure to pesticide applicators and field workers. It is estimated that a worker can absorb 6.0 mg of parathion from cotton 24 hours post-treatment with parathion and 3.0 mg 48 hours post-treatment during an actual 5-hour work day field exposure (Ware et al. 1973). Mean parathion residues extracted from the hands, shirts, and pants of four workers following a 5-hour field exposure in cotton fields 24 hours after application were 0.25, 6.1, and 9.8 mg, respectively, while estimated respiratory exposure during that time was 19.2 μg (mean air concentration of 3.2 ng/L) (Ware et al. 1974). Carman et al. (1982) reported mean

Table 6-5. Geometric Mean and Selected Percentiles of Urine Concentrations of Urinary *p*-Nitrophenol (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES IV)[a,b]

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) 50th	75th	90th	95th	Sample size
Total	1999–2000	*	<LOD	<LOD	2.50 (1.40–4.50)	5.00 (2.90–11.0)	1,989
	2001–2002	*	<LOD	1.33 (1.20–1.46)	2.69 (2.39–3.01)	3.72 (3.46–4.15)	2,975
	2007–2008	0.673 (0.595–0.761)	0.740 (0.660–0.830)	1.49 (1.32–1.66)	2.77 (2.19–3.45)	4.50 (3.50–5.42)	2,564
	2009–2010	0.454 (0.407–0.506)	0.510 (0.440–0.580)	1.09 (1.00–1.19)	2.18 (1.99–2.34)	3.14 (2.85–3.55)	2,744
Age group							
6–11 years	1999–2000	*	<LOD	0.940 (<LOD–2.40)	2.67 (1.70–3.80)	4.30 (2.70–6.40)	479
	2001–2002	*	0.790 (<LOD–0.910)	1.49 (1.36–1.61)	2.89 (2.22–3.58)	4.10 (3.01–4.74)	565
	2007–2008	0.803 (0.678–0.952)	0.890 (0.760–1.01)	1.66 (1.26–1.99)	2.85 (2.10–3.94)	4.37 (2.91–6.75)	383
	2009–2010	0.506 (0.426–0.601)	0.600 (0.500–0.720)	1.20 (0.980–1.43)	2.21 (1.74–2.73)	2.85 (2.28–3.81)	386
12–19 years	1999–2000	*	<LOD	<LOD	3.40 (1.60–5.70)	5.70 (2.60–19.0)	680
	2001–2002	*	0.730 (<LOD–0.910)	1.45 (1.32–1.61)	2.66 (2.15–3.11)	3.34 (3.11–4.01)	813
	2007–2008	0.769 (0.614–0.962)	0.850 (0.680–1.02)	1.49 (1.28–1.74)	2.79 (1.94–3.45)	3.47 (2.97–4.48)	387
	2009–2010	0.430 (0.375–0.493)	0.520 (0.460–0.590)	0.950 (0.870–1.09)	1.84 (1.43–2.03)	2.37 (1.84–2.98)	401
20–59 years	1999–2000	*	<LOD	<LOD	2.30 (1.20–5.70)	4.50 (2.30–16.0)	830
	2001–2002	*	<LOD	1.28 (1.09–1.47)	2.69 (2.32–3.10)	3.72 (3.37–4.24)	1,099
	2007–2008	0.658 (0.574–0.754)	0.720 (0.640–0.840)	1.49 (1.31–1.65)	2.77 (2.10–3.70)	4.68 (3.37–5.56)	1,173
	2009–2010	0.452 (0.400–0.511)	0.510 (0.420–0.590)	1.12 (1.00–1.24)	2.16 (1.91–2.39)	3.27 (2.84–3.58)	1,308
≥60 years	2001–2002	*	<LOD	1.29 (1.07–1.49)	2.66 (2.11–3.39)	4.01 (3.17–7.19)	498
	2007–2008	0.607 (0.512–0.720)	0.610 (0.550–0.710)	1.41 (1.14–1.76)	2.81 (2.19–3.90)	4.70 (2.90–6.91)	621
	2009–2010	0.453 (0.386–0.530)	0.460 (0.380–0.580)	1.06 (0.970–1.33)	2.42 (1.87–3.00)	3.65 (3.00–4.36)	649

Table 6-5. Geometric Mean and Selected Percentiles of Urine Concentrations of Urinary *p*-Nitrophenol (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES IV)[a,b]

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)				Sample size
			50th	75th	90th	95th	
Gender							
Males	1999–2000	*	<LOD	<LOD	2.50 (1.40–4.50)	4.50 (2.50–14.0)	971
	2001–2002	*	0.760 (0.450–0.880)	1.49 (1.32–1.63)	3.01 (2.66–3.33)	4.13 (3.61–4.92)	1,395
	2007–2008	0.782 (0.690–0.887)	0.850 (0.740–0.980)	1.59 (1.43–1.73)	2.85 (2.19–3.53)	4.52 (3.47–5.01)	1,282
	2009–2010	0.524 (0.472–0.581)	0.590 (0.490–0.670)	1.30 (1.14–1.42)	2.29 (2.07–2.54)	3.29 (2.90–3.73)	1,342
Females	1999–2000	*	<LOD	<LOD	2.50 (1.30–5.70)	5.70 (2.90–9.50)	1,018
	2001–2002	*	<LOD	1.18 (1.01–1.37)	2.29 (1.95–2.69)	3.52 (3.16–3.77)	1,580
	2007–2008	0.582 (0.510–0.664)	0.640 (0.550–0.720)	1.32 (1.10–1.59)	2.72 (2.16–3.35)	4.37 (3.09–5.64)	1,282
	2009–2010	0.396 (0.352–0.446)	0.440 (0.380–0.510)	0.960 (0.860–1.06)	2.01 (1.70–2.26)	3.07 (2.62–3.55)	1,402
Race/ethnicity							
Mexican Americans	1999–2000	*	<LOD	1.70 (<LOD–3.50)	5.80 (2.60–24.0)	22.0 (3.60–36.0)	695
	2001–2002	*	0.680 (<LOD–0.840)	1.33 (1.08–1.58)	2.61 (1.91–3.41)	3.64 (2.70–5.73)	744
	2007–2008	0.624 (0.542–0.720)	0.700 (0.560–0.810)	1.37 (1.16–1.52)	2.58 (2.03–3.33)	4.46 (2.79–6.91)	494
	2009–2010	0.484 (0.392–0.599)	0.560 (0.440–0.710)	1.30 (1.03–1.46)	2.21 (1.78–2.46)	3.07 (2.39–3.73)	602
Non–Hispanic blacks	1999–2000	*	<LOD	1.20 (<LOD–2.60)	2.90 (1.70–6.00)	4.80 (2.50–9.20)	518
	2001–2002	*	0.850 (<LOD–1.10)	1.76 (1.36–2.15)	3.13 (2.47–4.26)	4.92 (3.75–6.36)	752
	2007–2008	0.826 (0.716–0.952)	0.860 (0.760–1.01)	1.71 (1.45–1.92)	3.15 (2.56–3.90)	4.72 (3.91–5.68)	568
	2009–2010	0.505 (0.381–0.670)	0.570 (0.400–0.820)	1.30 (1.01–1.50)	2.19 (1.80–2.63)	3.49 (2.57–4.28)	504
Non–Hispanic whites	1999–2000	*	<LOD	<LOD	2.10 (<LOD–6.33)	4.20 (2.10–11.0)	603
	2001–2002	*	<LOD	1.29 (1.14–1.42)	2.70 (2.38–3.10)	3.71 (3.38–4.00)	1,259
	2007–2008	0.623 (0.531–0.730)	0.690 (0.610–0.790)	1.36 (1.19–1.59)	2.51 (1.89–3.08)	3.63 (2.82–5.48)	1,075
	2009–2010	0.440 (0.388–0.499)	0.490 (0.410–0.580)	1.03 (0.930–1.12)	2.18 (1.89–2.48)	3.14 (2.67–3.62)	1,197

[a] *p*-Nitrophenol is not unique to parathion as it is also a metabolite of methyl parathion and nitrobenzene.
[b] Note that *p*-nitrophenol is also a metabolite of methyl parathion and nitrobenzene.

CI = confidence interval; LOD = limit of detection

Source: CDC 2015

Table 6-6. Geometric Mean and Selected Percentiles of Urine Concentrations of Urinary *p*-Nitrophenol (Creatinine Corrected) (in µg/g) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES IV)[a]

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)				Sample size
			50th	75th	90th	95th	
Total	1999–2000	*	<LOD	<LOD	2.08 (1.33–3.91)	4.25 (2.15–10.2)	1,989
	2001–2002	*	<LOD	0.987 (0.868–1.10)	1.97 (1.86–2.13)	2.98 (2.60–3.33)	2,973
	2007–2008	0.692 (0.632–0.757)	0.673 (0.624–0.737)	1.28 (1.15–1.40)	2.38 (2.11–2.66)	3.57 (2.94–4.39)	2,562
	2009–2010	0.473 (0.430–0.521)	0.490 (0.438–0.531)	0.923 (0.851–1.01)	1.87 (1.60–2.06)	2.62 (2.32–2.91)	2,744
Age group							
6–11 years	1999–2000	*	<LOD	0.938 (<LOD–1.95)	2.80 (1.94–4.00)	4.20 (3.33–6.70)	479
	2001–2002	*	0.715 (<LOD–0.870)	1.60 (1.30–1.82)	2.78 (2.31–3.11)	3.67 (3.11–4.61)	565
	2007–2008	1.02 (0.892–1.17)	1.02 (0.920–1.12)	1.67 (1.46–1.90)	3.13 (2.51–3.64)	3.76 (3.31–4.89)	383
	2009–2010	0.684 (0.598–0.784)	0.717 (0.650–0.826)	1.30 (1.10–1.54)	2.19 (2.00–2.59)	3.16 (2.33–3.67)	386
12–19 years	1999–2000	*	<LOD	<LOD	1.80 (1.08–3.04)	4.00 (1.57–7.29)	680
	2001–2002	*	0.373 (<LOD–0.503)	0.840 (0.790–0.951)	1.59 (1.37–1.78)	2.10 (1.78–2.43)	812
	2007–2008	0.597 (0.511–0.698)	0.581 (0.497–0.697)	1.02 (0.891–1.14)	1.64 (1.29–2.17)	2.92 (1.55–4.54)	385
	2009–2010	0.368 (0.319–0.426)	0.413 (0.350–0.488)	0.684 (0.615–0.732)	1.09 (0.930–1.48)	1.73 (1.47–2.09)	401
20–59 years	1999–2000	*	<LOD	<LOD	2.00 (1.17–4.25)	4.29 (2.13–12.3)	830
	2001–2002	*	<LOD	0.875 (0.693–1.07)	1.79 (1.56–2.05)	2.89 (2.35–3.33)	1,099
	2007–2008	0.656 (0.595–0.724)	0.635 (0.585–0.706)	1.22 (1.09–1.34)	2.19 (1.94–2.51)	3.04 (2.56–4.07)	1,173
	2009–2010	0.452 (0.410–0.498)	0.453 (0.405–0.521)	0.868 (0.805–0.981)	1.61 (1.48–1.87)	2.34 (2.08–2.67)	1,308
≥60 years	2001–2002	*	<LOD	1.21 (.920–1.56)	2.29 (1.99–2.83)	4.29 (2.51–5.67)	497
	2007–2008	3.26 (2.46–4.47)	0.763 (0.638–0.909)	1.59 (1.40–1.73)	0.755 (0.643–0.886)	5.43 (3.33–7.09)	621
	2009–2010	0.537 (0.450–0.641)	0.520 (0.451–0.636)	1.15 (.899–1.59)	2.53 (2.03–3.39)	4.27 (2.98–5.26)	649

Table 6-6. Geometric Mean and Selected Percentiles of Urine Concentrations of Urinary *p*-Nitrophenol (Creatinine Corrected) (in µg/g) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES IV)[a]

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)				Sample size
			50th	75th	90th	95th	
Gender							
Males	1999–2000	*	<LOD	<LOD	1.90 (1.01–3.39)	3.39 (1.77–7.55)	971
	2001–2002	*	0.984 (0.869–1.07)	1.89 (1.66–2.09)	0.433 (0.333–0.534)	2.98 (2.29–3.57)	1,395
	2007–2008	0.650 (0.595–0.712)	0.641 (0.594–0.697)	1.20 (1.08–1.33)	2.11 (1.63–2.49)	3.02 (2.51–3.71)	1,281
	2009–2010	0.466 (0.424–0.512)	0.495 (0.432–0.538)	0.922 (0.843–0.985)	1.84 (1.48–2.05)	2.36 (2.21–2.68)	1,342
Females	1999–2000	*	<LOD	<LOD	2.26 (1.48–4.88)	6.92 (2.76–14.1)	1,018
	2001–2002	*	<LOD	0.995 (0.801–1.23)	2.08 (1.85–2.32)	3.04 (2.58–3.44)	1,578
	2007–2008	0.735 (0.660–0.818)	0.707 (0.644–0.779)	1.37 (1.21–1.63)	2.67 (2.20–3.25)	4.11 (3.11–5.06)	1,281
	2009–2010	0.480 (0.430–0.536)	0.488 (0.426–0.538)	0.923 (0.824–1.08)	1.93 (1.62–2.17)	2.72 (2.32–3.46)	1,402
Race/ethnicity							
Mexican Americans	1999–2000	*	<LOD	1.55 (<LOD–3.17)	4.86 (2.21–21.9)	17.4 (3.94–47.7)	695
	2001–2002	*	0.389 (<LOD–0.546)	0.935 (0.716–1.22)	1.89 (1.43–2.63)	3.23 (2.49–3.84)	744
	2007–2008	0.638 (0.538–0.757)	0.604 (0.554–0.708)	1.14 (0.938–1.40)	2.03 (1.45–3.57)	3.63 (2.05–4.71)	493
	2009–2010	0.506 (0.419–0.612)	0.541 (0.480–0.621)	1.02 (0.854–1.19)	1.73 (1.39–2.15)	2.35 (1.84–3.02)	602
Non–Hispanic blacks	1999–2000	*	<LOD	0.683 (<LOD–1.79)	2.07 (1.33–3.71)	3.71 (1.98–7.20)	518
	2001–2002	*	0.438 (<LOD–0.640)	1.04 (0.847–1.27)	1.84 (1.59–2.30)	2.81 (2.14–4.30)	751
	2007–2008	0.633 (0.565–0.708)	0.634 (0.553–0.743)	1.07 (0.973–1.23)	2.04 (1.64–2.32)	2.66 (2.11–3.54)	567
	2009–2010	0.382 (0.289–0.505)	0.431 (0.312–0.554)	0.785 (0.671–0.874)	1.36 (1.03–1.99)	2.52 (1.73–3.22)	504
Non–Hispanic whites	1999–2000	*	<LOD	<LOD	1.97 (<LOD–4.29)	3.83 (1.97–10.2)	603
	2001–2002	*	<LOD	0.984 (0.827–1.16)	2.06 (1.89–2.29)	3.11 (2.49–3.57)	1,258
	2007–2008	0.677 (0.604–0.759)	0.664 (0.604–0.761)	1.25 (1.10–1.38)	2.28 (2.00–2.56)	3.23 (2.60–4.00)	1,075
	2009–2010	0.479 (0.433–0.530)	0.490 (0.433–0.538)	0.923 (0.849–1.03)	1.92 (1.60–2.13)	2.62 (2.24–3.04)	1,197

[a] *p*-Nitrophenol is not unique to parathion as it is also a metabolite of methyl parathion and nitrobenzene.

CI = confidence interval; LOD = limit of detection

Source: CDC 2015

levels of parathion ranging from <0.01 to 7.5 μg/cm^2/hour deposited on test patches on the bodies of workers while using airblast spray equipment and different formulations of parathion to spray citrus trees.

Before the use of parathion was banned, it was found to be transported into homes by translocation of vapors and by track-in from outdoors on shoes and clothing from workers. There have been studies showing that the air inside and outside the homes of workers that used parathion contained measurable amounts of parathion, with inside air containing higher concentrations (IARC 1983), which would be a source of exposure to family members. The Colorado Community Pesticide Sampling Programs detected parathion in monthly air samples taken inside and outside the homes of men occupationally exposed to parathion, with 64 of 94 and 44 of 87 samples testing positive, respectively (Tessari and Spencer 1971).

NIOSH recommends that the occupational exposure level not exceed 0.05 mg/m^3 for a 10-hour TWA workday (NIOSH 2013). In addition, the American Conference of Governmental Industrial Hygienists has recommended a time-weighted average threshold limit value (TWA-TLV) of 0.05 mg/m^3 with an inhalable, vapor, skin notation for occupational exposure to parathion (NIOSH 2013).

The National Occupational Hazard Survey (NOHS) conducted by NIOSH in 1974 estimated that 302 workers employed at 43 facilities were potentially exposed to parathion in the United States (NIOSH 2013). The NOHS survey did not contain information on the frequency, concentration, or duration of exposure; the survey provided only estimates of workers potentially exposed to chemicals in the workplace. Since parathion is no longer registered for use in the United States, there is no updated information in regards to worker exposure in this country.

6.6 EXPOSURES OF CHILDREN

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

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

Children can be exposed to pesticides from multiple sources and pathways. In addition to the standard pathways of diet and drinking water, children in agricultural communities may be exposed through farm proximity and parental take-home from agricultural use (Fenske et al. 2000).

Children are not likely to be exposed to parathion and its residues in the foods that they eat since parathion has not been made or used substantially in the United States for several years. A good indicator of general public exposure to a substance is the National Report on Human Exposure to Environmental Chemicals. Concentrations of the parathion metabolite, *p*-nitrophenol, were low in the urine of children and all other segments of the population during the initial 1999–2000 and final 2001–2002 sampling periods; however, these detections may not be attributed solely to parathion since *p*-nitrophenol is also a metabolite to several other organophosphorus pesticides, including methyl parathion (CDC 2005). Since then, it is likely that the levels of this metabolite in children have decreased.

In the FDA Total Diet Study for infants and toddlers conducted between 1980 and 1982, the average concentration and the calculated average daily intake of parathion in different food groups were determined (Gartrell et al. 1986a). In the infant diet, parathion was only detected in fruit and fruit juices at an average concentration (mg/kg) and average daily intake (μg/day) of 0.0002 and 0.0211, respectively, while none was detected in the other food groups. In the toddler diet, the average concentrations (mg/kg) and average daily intakes (μg/day) of parathion by food group were 0.0001 and 0.0080 in vegetables and 0.0002 and 0.0147 in fruit and fruit juices, respectively. No parathion was detected in drinking water for either study groups, although the LOD was not reported. Data on the weight-adjusted intake of parathion by infants and toddlers were determined based on the results of the FDA Total Diet Studies for fiscal years 1978–1981/1982 (Gartrell et al. 1986a). The reported weight-adjusted intakes of parathion ranged from 0.002 to 0.005 μg/kg body weight/day for infants and from not detected to 0.003 μg/kg body weight/day for toddlers for the study years.

In 1982–1984, a national U.S. FDA Total Diet Study was performed that showed the average daily intakes of parathion in the United States for children 6–11 months of age and 2 years of age were 11.2 and 5.0 ng/kg body weight/day, respectively (Gunderson 1988). In the FDA Total Diet Study performed between July 1986 and April 1991, the average daily intakes of parathion in the U.S. for children 6–11 months of age and 2 years of age were 0.008 and 0.004 μg/kg body weight/day, respectively (Gunderson 1995).

Children of agricultural families are likely exposed to agricultural chemicals, even if they are not involved in farm activities. Simcox et al. (1995) designed a study to determine whether such children were exposed to higher levels of pesticides, including parathion, than children whose parents were not involved in agriculture and whose homes were not close to farms. Household dust and soil samples were collected from children's play areas from 59 residences in eastern Washington State (26 farming, 22 farmworkers, and 11 non-farming families) during the 1992 spray season. The study reported that parathion was detected in yard soil samples collected from farm families, a majority living within 200 feet of an operating apple or pear orchard, at concentrations ranging from not detected to 932 ng/g, with a mean concentration of 46 ng/g, while none was detected in non-farm reference family's yard soil. In household dust samples, parathion was detected at concentrations ranging from below the limit of quantitation (LOQ) to 2,786 ng/g (mean of 365 ng/g) in agricultural families homes and <LOQ to 425 ng/g (mean of 76 ng/g) in non-farm family homes. Household dust concentrations were significantly lower in reference homes when compared to farmer/farmworker homes, demonstrating that children of agricultural families have a higher potential for exposure to parathion than children of non-farm families.

6.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Since parathion has not been used substantially in the United States for several years and parathion residues in the environment from historical spraying would be degraded at present, there are likely to be no populations in the United States with potentially high exposures to this substance. According to the TRI, small amounts of unused product are occasionally transported for disposal at landfills or hazardous waste incinerators, but this represents a small population of workers who may be exposed to low quantities of this substance. In the past, pesticide formulators and applicators employed in facilities that manufactured parathion, and farmers were exposed to levels greater than the general population.

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 parathion is available. Where adequate information is not available, ATSDR, in conjunction with NTP, is required to assure the initiation of a program of research designed to determine the health effects (and techniques for developing methods to determine such health effects) of parathion.

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-chemical properties of parathion are provided in Chapter 4. Important properties such as melting point, boiling point, vapor pressure, solubility, log K_{ow} and Henry’s Law constant are available. No data needs are identified.

Production, Import/Export, Use, Release, and Disposal. According to the Emergency Planning and Community Right-to-Know Act of 1986, 42 U.S.C. Section 11023, industries are required to submit substance release and off-site transfer information to the EPA. The TRI, which contains this information for 2014, became available in February of 2015. This database is updated yearly and should provide a list of industrial production facilities and emissions. Since parathion is not registered for use any longer in the United States, it is unlikely to be produced in any significant quantities. According to data from the TRI, occasionally small quantities of unused stockpiles are transported to hazardous waste facilities for disposal.

Environmental Fate. The environmental fate and transport of parathion is understood and no data needs are identified. The mobility of parathion in soils is expected to be low based on measured K_{oc} values. Volatilization is generally considered low. Hydrolysis, photolysis, and biodegradation account for the removal of parathion from the environment. Sorption of parathion to organic matter in natural waters and soils affects its susceptibility for aquatic photolysis, its bioavailability for aquatic organisms, and its biodegradation (Schuurmann et al. 2006). It follows two major fate pathways: degradation to less toxic compounds or oxidative conversion to the toxic bioactive product, paraoxon (CDFA 1988).

Bioavailability from Environmental Media. Parathion has been detected in aquatic and terrestrial organisms (HSDB 2013) and is, therefore, bioavailable to some extent from environmental media.

Adsorption to organic matter in soil and sediment reduces its bioavailability (Schuurmann et al. 2006). No data needs are identified.

Food Chain Bioaccumulation. Measured BCFs of parathion in fish suggest that bioaccumulation in aquatic organisms is not high. A better understanding of the biochemical and physiological basis of the processes of parathion uptake, biotransformation, and excretion is needed in order to fully understand species differences in parathion metabolism in fish and shellfish. The use of molecular biology in obtaining further knowledge about the enzymes and transport proteins important in pesticide metabolism should provide further advances in this field. This will be useful for predicting the likelihood that parathion residue will remain in edible parts of food-producing fish and shellfish (James 1994) if the future use of parathion is authorized or if it is used in countries other than the United States.

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

Exposure Levels in Humans. When parathion was used, humans were exposed to it by inhalation of air and intake from food and drinking water. Since parathion is no longer used substantially in the United States, exposure to humans is expected to be low. Future research to assess the potential for parathion exposure due to contaminated hazardous waste sites would be useful since it has been detected in 20 NPL sites (ATSDR 2015).

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

Exposures of Children. Parathion’s former use in the United States may have led to low levels of exposure to children; however, since the last uses of parathion were cancelled more than a decade ago, current exposure is considered low. No data needs are identified.

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 parathion were located. This substance is not currently one of the compounds for which a sub-registry has been established in the National Exposure Registry. The substance will be considered in the future when chemical selection is made for sub-registries to be established. The information that is amassed in the National Exposure Registry facilitates the epidemiological research needed to assess adverse health outcomes that may be related to exposure to this substance.

6.8.2 Ongoing Studies

Since parathion is no longer produced or used substantially in the United States (EPA 2000), no ongoing studies regarding its environmental fate or physical properties are being performed in this country. However, foreign studies regarding parathion continue to be published.