6. POTENTIAL FOR HUMAN EXPOSURE

6.1 OVERVIEW

Di-*n*-butyl phthalate has been identified in at least 471 of the 1,585 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HazDat 2001). However, the number of sites evaluated for di-*n*-butyl phthalate is not known. The frequency of these sites can be seen in Figure 6-1. Of these sites, 469 are located within the United States and 2 are located in the Commonwealth of Puerto Rico (not shown).

Di-*n*-butyl phthalate is one of several phthalate esters that are widely used as a plasticizers in a range of polymers (e.g., polyvinyl acetate, nitrocellulose) that are found in many common consumer products including home furnishings, paints, clothing, and cosmetic products. Because of its many uses, di-*n*-butyl phthalate is widespread in the environment and has been identified at low levels in all environmental media. Therefore, humans may be exposed to di-*n*-butyl phthalate by inhalation, by ingestion of water or food containing di-*n*-butyl phthalate, and by dermal contact with plastics, cosmetics, or other materials containing di-*n*-butyl phthalate.

In air, di-*n*-butyl phthalate may be adsorbed to particulate matter or occur as a vapor. Di-*n*-butyl phthalate is expected to decompose in the air, or be transported to water and/or soil by wet (snow or rain) or dry (wind and settling) deposition. Di-*n*-butyl phthalate is taken up from water by a variety of aquatic organisms. In water and soil, di-*n*-butyl phthalate is subject to microbial degradation; degradation in the presence (aerobic) and absence (anaerobic) of oxygen has been reported. Exposure of the general population to di-*n*-butyl phthalate may occur through contact with contaminated air, water, food, soil, and/or products in which di-*n*-butyl phthalate is intentionally incorporated.

6.2 RELEASES TO THE ENVIRONMENT

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

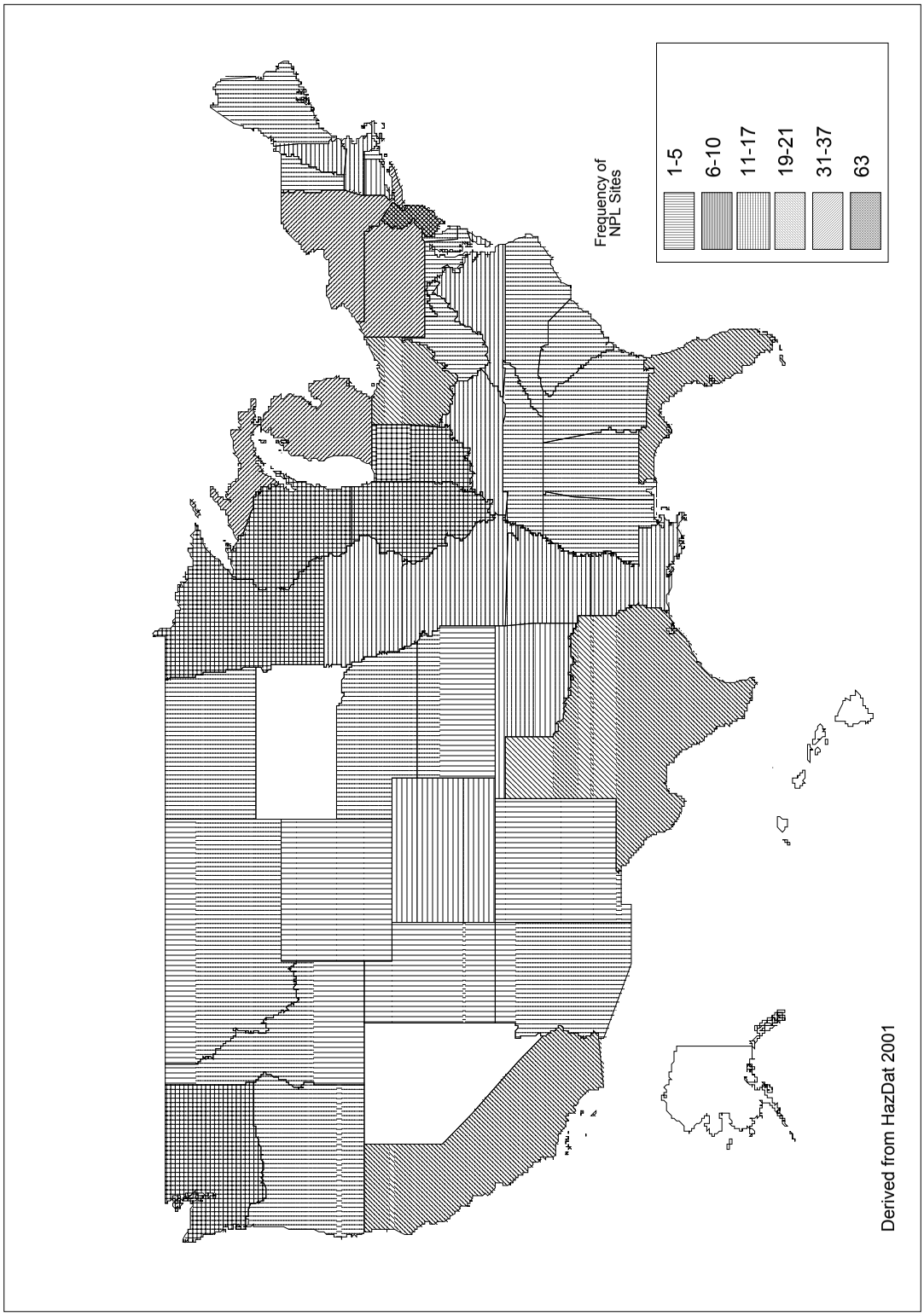

Figure 6-1. Frequency of NPL Sites with Di-n-butyl Phthalate Contamination

Di-*n*-butyl Phthalate has been identified in a variety of environmental media (air, surface water, leachate, groundwater, soil, and sediment) collected at 471 of 1,585 current or former NPL hazardous waste sites (HazDat 2001).

6.2.1 Air

As presented in Chapter 4, di-*n*-butyl phthalate has a relatively low vapor pressure and Henry's law constant, as well as a relatively high octanol/water partition coefficient and soil sorption coefficient. This combination of properties is consistent with a chemical that is found to only a limited extent in air (see Staples et al. 1997). However, di-*n*-butyl phthalate appears to be ubiquitous in air at very low levels (see Section 6.4), apparently from the widespread distribution and use of plastics and other products containing di-*n*-butyl phthalate. In Sweden, Thurén and Larsson (1990) could not correlate di-*n*-butyl phthalate fallout rates with specific sources or transport routes on a nationwide basis. They found no "distributional patterns or gradient," which suggests that any local patterns were obscured by di-*n*-butyl phthalate contribution from other sources or that emission sources of roughly equal magnitude were diffuse. In contrast, monitoring data show that elevated fallout concentrations of bis(2-ethylhexyl) phthalate (a similar molecule) are associated with industrial activity (Thurén and Larsson 1990). Elevated fallout concentrations of bis(2-ethylhexyl) phthalate were only seen near a stack, and no elevated concentrations could be seen 2 km away from the stack. No pattern associating distance from sources and concentration was seen with di-*n*-butyl phthalate by Ritsema et al. (1989) in Lake Yssel in the Netherlands, while for other higher-molecular-weight phthalate esters, a pattern was evident. The authors suggested that atmospheric fallout from diffuse sources may be the dominant mechanism by which di-*n*-butyl phthalate enters the lake.

The possibility of many diffuse sources of di-*n*-butyl phthalate is supported by some of the uses. For example, some of the products that contain di-*n*-butyl phthalate include thin sheets and coatings (including such products as floor wax [Weschler et al. 1990] and paint [Crump 1995]). These products characteristically have large surface area-to-volume ratios, which may allow di-*n*-butyl phthalate to volatilize more readily relative to other products with smaller surface area-to-volume ratios. For example, Crump (1995) reported that 0.04 mg/m^3 of di-*n*-butyl phthalate, which had volatilized from paint, was detected in indoor air. Cadogan and Howick (1996) reported an indoor overall emission rate of 2.3×10^{-4} mg/second-m^3 at 25 EC for all phthalate plasticizers in products such as wall coverings, flooring, upholstery, and wire insulation.

Di-*n*-butyl phthalate is a component of cigarette and wood smoke (Otson et al. 1991) and waste incinerator stacks (Jay and Stieglitz 1995; Nishikawa et al. 1992). Di-*n*-butyl phthalate is listed by EPA as an inert ingredient in some pesticides (EPA 1999a) and an active ingredient in others (EPA 1999b).

Releases to the atmosphere have been reported in EPA Toxic Release Inventory (TRI99 2001), and the quantities are presented in Table 6-1. The TRI data should be used with caution because only certain types of facilities are required to report. This is not an exhaustive list. Nonetheless, according to TRI99 (2001), manufacturing and processing facilities in 32 states emitted an estimated 36,477 pounds (*ca.* 17 metric tons) to the air, or approximately 9% of the total environmental release.

Di-*n*-butyl phthalate has also been identified in the condensed water (Jay and Stieglitz 1995) and exhaust gas (Nishikawa et al. 1992) from waste incinerators. In addition, Jones et al. (1996) used the CORAL+ model to estimate that a large U.S. city would emit approximately 200 g/year from all manholes, based on sewage concentrations of 7 µg di-*n*-butyl phthalate/L of sewage. The model was very sensitive to manhole ventilation rates and the authors noted that the estimate was probably very conservative. Di-*n*-butyl phthalate has been identified in 2 air samples collected from the 471 NPL hazardous waste sites where it was detected in some environmental media (HazDat 2001).

6.2.2 Water

Bauer and Herrmann (1997) reported the concentration of di-*n*-butyl phthalate in the leachate from various fractions of household wastes from the regions of Bayreuth and Straubling in Germany. The wastes included food waste, paper for recycling, unusable paper, cardboard, plastic films, other plastics, textiles, 8–40 mm screened fraction, <8 mm screened fraction, compound packing waste, compound materials, and disposable diapers. It is anticipated that household waste from continental Europe is similar to that in the United States, so the same profile would be expected in both places. Further information on this study is presented in Section 6.4.

According to TRI99 (2001), releases to water amount to approximately 279 pounds (0.13 metric tons) or approximately 0.07% of the total amount released to all compartments (see Table 6-1). Releases to underground injection wells amount to approximately 290,000 pounds (132 metric tons) or approximately 74% of the total amount released to all compartments. These releases occurred only in the

Table 6-1. Releases to the Environment from Facilities that Produce, Process, or Use Di-n-butyl Phthalate

		Reported amounts released in pounds per year[a]						
State[b]	Number of facilities	Air[c]	Water	Underground injection	Land	Total on-site release[d]	Total off-site release[e]	Total on- and off-site release
AL	1	No data	No data	No data	No data	No data	No data	No data
AZ	1	192	No data	No data	No data	192	No data	192
AR	4	751	0	No data	No data	751	No data	751
CA	14	568	0	No data	No data	568	1,324	1,892
CO	2	0	0	No data	No data	0	No data	0
FL	4	910	0	290,000	No data	290,910	250	291,160
GA	5	575	No data	No data	No data	575	No data	575
IL	16	4,070	0	No data	No data	4,070	20,722	24,792
IN	6	34	No data	No data	No data	34	1,127	1,161
IA	4	1	No data	No data	No data	1	171	172
KS	2	2,360	No data	No data	No data	2,360	2,069	4,429
KY	2	0	No data	No data	No data	0	No data	0
LA	2	0	0	No data	No data	0	No data	0
MD	1	0	No data	No data	No data	0	No data	0
MA	8	251	No data	No data	No data	251	No data	251
MI	6	43	0	No data	9,421	9,464	9,718	19,182
MN	2	No data	No data	No data	No data	No data	No data	No data
MO	6	501	No data	No data	No data	501	318	819
NE	1	10	No data	No data	No data	10	1,014	1,024
NJ	18	415	0	No data	0	415	4	419
NY	10	267	92	No data	0	359	1,031	1,390
NC	12	3,862	0	No data	4,224	8,086	8,662	16,748
OH	13	22	1	No data	No data	23	1,390	1,413
OK	2	5	No data	No data	No data	5	No data	5
PA	6	260	No data	No data	No data	260	No data	260

Table 6-1. Releases to the Environment from Facilities that Produce, Process, or Use Di-n-butyl Phthalate (*continued*)

State[b]	Number of facilities	Reported amounts released in pounds per year[a]						
		Air[c]	Water	Underground injection	Land	Total on-site release[d]	Total off-site release[e]	Total on- and off-site release
RI	1	365	No data	No data	No data	365	1,800	2,165
TN	4	14,211	160	No data	No data	14,371	No data	14,371
TX	14	436	0	No data	0	436	No data	436
UT	1	13	No data	No data	No data	13	No data	13
VA	5	851	21	No data	No data	872	No data	872
WA	2	5,498	5	No data	0	5,503	250	5,753
WI	5	6	No data	No data	No data	6	500	506
Total	184	36,477	279	290,000	13,645	340,401	50,350	390,751

Source: TRI99 2001

[a]Data in TRI are maximum amounts released by each facility.
[b]Post office state abbreviations are used.
[c]The sum of fugitive and stack releases are included in releases to air by a given facility.
[d]The sum of all releases of the chemical to air, land, water, and underground injection wells.
[e]Total amount of chemical transferred off-site, including to publicly owned treatment works (POTW).

state of Florida. The TRI data should be used with caution because only certain types of facilities are required to report. This is not an exhaustive list. Di-*n*-butyl phthalate has been identified in 217 groundwater and 78 surface water samples collected form the 471 NPL hazardous waste sites, where it was detected in some environmental media (HazDat 2001).

Di-*n*-butyl phthalate may also be released into surface waters from industrial sources (Sheldon and Hites 1979), municipal waste water (Stubin et al. 1996), and leachate from sanitary landfills (EPA 1990a; Kinman et al. 1995). Di-*n*-butyl phthalate has also been reportedly released to groundwater from a hazardous waste site (Eckel et al. 1993).

Di-*n*-butyl phthalate has been detected in 5% of the urban runoff samples from 2 of the 19 cities tested by EPA (Howard 1989). Concentrations in this urban waste water ranged from 0.5 to 11 µg/L. Sewage sludge has been shown to concentrate di-*n*-butyl phthalate from water by about 25-fold. A concentration of 966 µg/L sludge was reported by Feiler et al. (1980). It is believed that di-*n*-butyl phthalate in waste water comes from a variety of sources including wash water used on vinyl flooring (Cadogan and Howick 1996). Disposal of secondary sewage effluent by rapid infiltration into the subsurface has been reported to produce a plume over 3,500 m long of groundwater contaminated with di-*n*-butyl phthalate at concentrations up to 450 mg/L (Barber 1992; Barber et al. 1988). Di-*n*-butyl phthalate has been identified in river and ocean sediments at points of sewage outflow from urban areas (Fallon and Horvath 1985; Swartz et al. 1983).

6.2.3 Soil

According to TRI99 (2001), releases to land amount to approximately 13,645 pounds (6.2 metric tons) or approximately 3.5% of the total (see Table 6-1). Based on this analysis of TRI data alone, soil does not appear to be a medium that commonly receives industrial releases. The TRI data should be used with caution because only certain types of facilities are required to report. This is not an exhaustive list. Di-*n*-butyl phthalate has been identified in 280 soil and 126 sediment samples collected from the 471 NPL hazardous waste sited where it was detected in some environmental media (HazDat 2001).

No other release of di-*n*-butyl phthalate to soils has been reported; however, di-*n*-butyl phthalate has been reported in leachate from sanitary landfills (EPA 1990a; Kinman et al. 1995) and in groundwater from a hazardous waste site (Eckel et al. 1993). These monitoring data indicate release to soil. Thurén and

Larsson (1990) also noted that the average yearly atmospheric input to soil in Sweden was 202 µg/m^2 for di-*n*-butyl phthalate. Also, di-*n*-butyl phthalate may seep into soil from di-*n*-butyl phthalate containing sewage sludge that is deposited on land.

6.3 ENVIRONMENTAL FATE

6.3.1 Transport and Partitioning

Di-*n*-butyl phthalate appears to be ubiquitous in air at low levels, is in both the vapor phase and associated with particulates, and is subject to both wet (rain and snow) and dry (wind and settling) deposition on the Earth's surface. Eisenreich et al. (1981) calculated that wet and dry deposition of di-*n*-butyl phthalate into the five Great Lakes amounted to 47.7 metric tons per year, which corresponds to an average fallout rate of 16.2 µg/m^2 per month. Thurén and Larsson (1990) reported that the average fallout rate in Sweden for di-*n*-butyl phthalate was 16.8 µg/m^2 per month (the range was 1.98–335.5 µg/m^2 per month). Thurén and Larsson (1990) also estimated the median fallout concentration to be 36 ng/L. The authors also noted that for a related phthalate ester, bis(2-ethylhexyl) phthalate, fallout rate decreased with increasing distance (but only up to 2 km away) from a stack at a facility that used bis(2-ethylhexyl) phthalate (see Section 6.2). A similar behavior is likely for di-*n*-butyl phthalate. Nonetheless, the authors noted that no specific overall fallout patterns were observed. This is consistent with a diffuse source of di-*n*-butyl phthalate in the atmosphere. In addition, di-*n*-butyl phthalate has been found in 2 m deep Antarctic snow (Desideri et al. 1994) and in the snow in the Austrian Alps (Gröllert et al. 1997), suggesting that di-*n*-butyl phthalate can be transported for long distances so that di-*n*-butyl phthalate measured in one part of the world may have originated elsewhere. This transport is likely particle sorbed di-*n*-butyl phthalate because vapor phase di-*n*-butyl phthalate reacts rapidly with hydroxyl radicals in the atmosphere (see Section 6.3.2.1), while the particle-sorbed chemical does not react rapidly with hydroxyl radicals (di-*n*-butyl phthalate is associated with particulates at low temperatures). Atmospheric fallout is negatively correlated with temperature so that less di-*n*-butyl phthalate is subject to fallout in the summer than in the winter (Staples et al. 1997; Thurén and Larsson 1990). This is in keeping with a higher proportion of the atmospheric di-*n*-butyl phthalate in the vapor state in the warm summer and less in the cold winter.

In water, di-*n*-butyl phthalate adsorbs to suspended particles, but, in many cases, the majority remains dissolved. Based on log K_{oc} values of 3.14 (Russell and McDuffie 1986) and 4.17 (Sullivan et al. 1982),

di-*n*-butyl phthalate is anticipated to sorb relatively strongly to suspended particulates and sediments (see Swann et al. 1983); however, this is not always observed. Dissolved di-*n*-butyl phthalate is more readily advected and may be more available for biodegradation than particle-bound material. When dissolved in water, di-*n*-butyl phthalate may be associated with dissolved humic materials, potentially increasing the apparent water solubility by a small amount. Based on Henry's law constants of $4.5x10^{-6}$ (Roy 1994) and $8.83x10^{-7}$ (Staples et al. 1997) atm-m^3/mole, di-*n*-butyl phthalate is not expected to volatilize significantly from water to the atmosphere (Lyman 1982). Germain and Langlois (1988) and Staples et al. (1997) reported that 83.5–86% of the di-*n*-butyl phthalate was dissolved and 14–16.5% was bound to suspended material (the concentrations of suspended material were 3.0–7.83 mg/L) in Niagara and St. Lawrence Rivers. By contrast, Ritsema et al. (1989) reported that 29–57% of the di-*n*-butyl phthalate in Lake Yssel water (the Netherlands) was dissolved, while 43–71% was associated with suspended material. Matsuda and Schnitzer (1971) reported that di-*n*-butyl phthalate was slightly more soluble in water containing fulvic acids extracted from soil than in pure water, and that solubility increased with increasing fulvic acid concentration. Therefore, di-*n*-butyl phthalate may be present in higher dissolved concentrations in waters with high concentrations of humic materials than in waters with low concentrations of humic materials. Finally, in an estuary in the United Kingdom, Preston and Al-Omran (1986, 1989) and Al-Omran and Preston (1987) reported that 66–86% of the di-*n*-butyl phthalate was dissolved, while 14–34% was bound to suspended material, and that more adsorption occurs at higher salt concentration. The authors reported that suspended material averaged 1,524 mg/L.

No reports were located that detailed the sorption of di-*n*-butyl phthalate specifically to freshwater sediments; however, it is anticipated that the sorption to these sediments will be similar to that of suspended materials. Sullivan et al. (1982) reported that di-*n*-butyl phthalate is rapidly adsorbed from seawater onto marine sediment, but they did not report a K_{oc} so that comparison with other reports is difficult.

In soil, di-*n*-butyl phthalate is expected to have limited mobility based on its log K_{oc} (see Table 4-2). However, some migration to groundwater occurs, but may be limited to soils with low organic content soils or sites with high water tables. Russell and McDuffie (1986) reported a log K_{oc} of 3.14 in a soil in Broome County, New York. According to Swann et al. (1983), this K_{oc} is indicative of relatively low mobility, suggesting that di-*n*-butyl phthalate will have limited migration to groundwater. Nonetheless, Eckel et al. (1993) reported finding di-*n*-butyl phthalate in the groundwater under the Hipps Road landfill NPL site in Florida. This site was developed at a location with unconsolidated medium- to fine-grained

sand that allowed rapid movement of percolating water. Similarly, Barber (1992) reported finding di-*n*-butyl phthalate in the groundwater near rapid infiltration beds where secondary sewage effluent was disposed for over 50 years, and Kinman et al. (1995) reported the presence of di-*n*-butyl phthalate in sanitary landfill leachate.

In addition to the cases noted above, the presence of organic solvents in hazardous waste sites may increase the solubility of compounds such as di-*n*-butyl phthalate, thus increasing the amounts that may leach from the site into the subsoil and into groundwater. For example, 1-octanol-saturated water increases the solubility of di-*n*-butyl phthalate approximately six times its normal water solubility (Nyssen et al. 1987).

Finally, data indicate that di-*n*-butyl phthalate can partition from food and water into a variety of organisms. Studies using radioactively labeled di-*n*-butyl phthalate have shown accumulation of radioactivity in aquatic invertebrates (Sanders et al. 1973) and fish (Wofford et al. 1981). Most of the accumulated radioactivity is apparently in the form of the primary metabolite, mono-*n*-butyl phthalate (Howard 1989). Numerous experiments have shown that the accumulation of di-*n*-butyl phthalate in the aquatic and terrestrial food chain is limited by biotransformation (i.e., transformation of chemical compounds within a living system), which progressively increases with trophic level (Staples et al. 1997). In general, bioconcentration factors decrease for organisms that possess more advanced metabolic systems (Stalling et al. 1973). For example, the mean bioconcentration factors for algae, crustaceans, insects, and fish are 3,399, 662±229, 624±144, and 167 mg/g (wet weight), respectively (Staples et al. 1997).

In greenhouse studies, Shea et al. (1982) demonstrated dose-dependent partitioning of di-*n*-butyl phthalate from soils into corn, soybean, and wheat seedlings. For example, plant to soil bioconcentration factors (defined as the ratio of wet weight concentrations in plant and soil, respectively) were <0.002 for corn, indicating the limited bioaccumulation potential of di-*n*-butyl phthalate by terrestrial plants.

6.3.2 Transformation and Degradation

It should be noted that many of the studies reporting di-*n*-butyl phthalate biodegradation use radiolabeled material as the substrate and measure $^{14}CO_2$ as an indication of the extent of biodegradation. The reported percentage of evolved $^{14}CO_2$ indicates the amount of mineralization, but is not necessarily an indication of

the extent of degradation. In addition to oxidation to CO_2, microbial communities can also, for example, incorporate organic substrates into cellular materials, or produce simple degradation products or high molecular-weight materials. All of these processes contribute to the overall removal of the substrate, but are not indicated by the percentage of $^{14}CO_2$ reported.

6.3.2.1 Air

In air, di-*n*-butyl phthalate in vapor form would be expected to react with hydroxyl radicals since it has extractable hydrogens. An overall hydroxyl radical reaction rate of 9.28×10^{-12} cm^3/molecule-second was calculated using the AOPWin program (version 1.88, Meylan and Howard 1993). Assuming 1.5×10^6 •OH molecules/cm^3, the half-life of di-*n*-butyl phthalate vapor in air can be calculated to be approximately 14 daylight hours. For di-*n*-butyl phthalate adsorbed to airborne particles, the half-life may be considerably longer and may account for the long range transport of di-*n*-butyl phthalate, but this has not been studied.

6.3.2.2 Water

Many papers describing the biodegradation of di-*n*-butyl phthalate in environmental waters and waste water are available in the literature (Scholz et al. 1997; Staples et al. 1997). Di-*n*-butyl phthalate is frequently reported to degrade 50–100% aerobically within 1–28 days in both fresh and marine water, and anaerobically over 90% within 30 days in fresh water (Staples et al. 1997). In a river die-away test using water from three rivers in the Netherlands, Schouten et al. (1979) reported that 90% of the initial 50 µg/L di-*n*-butyl phthalate was removed in 3 days, while only 10% was lost in the sterile controls. Johnson et al. (1984) found that di-*n*-butyl phthalate was degraded aerobically in water and sediment from Little Dixie Lake in Columbia, Missouri. Approximately 63% degradation was observed in 7 days and 70% degradation was observed in 14 days as measured by $^{14}CO_2$ evolution using initial di-*n*-butyl phthalate concentrations of 0.082–8.2 mg/L. In fresh and estuarine waters from U.S. sources, EPA (1984a) reported that the range of time for one-half of the initial di-*n*-butyl phthalate (500 µg/L) to disappear was 1.7–13 days. The authors reported a 0–7 day lag period, and no degradation was seen in sterile controls. Johnson and Lulves (1975) reported that di-*n*-butyl phthalate (initial concentration of 1 mg/L) incubated anaerobically in sediment degraded 61% in 7 days and 98% in 30 days as measured by recovery of di-*n*-butyl phthalate.

In contrast to biodegradation, abiotic hydrolysis appears to be a slow process. EPA (1989d) reported that the hydrolysis half-life for di-*n*-butyl phthalate was approximately 22 years.

In screening tests using activated sludge, removal of >90% of the initial di-*n*-butyl phthalate is reported. For example, Kodama and Takai (1974) reported that 97–99.5% of the di-*n*-butyl phthalate at 0.1–1.0 mg/L degraded in a 3-day test. Using acclimated sludge in a semi-continuous activated sludge test, O'Grady et al. (1985) reported that >90% of the initial of 3 mg/L concentration of di-*n*-butyl phthalate degraded in 1 day. Under anaerobic conditions in activated sludge, di-*n*-butyl phthalate was completely degraded to carbon dioxide and methane over a period of 20 days (Hannah et al. 1986). Shelton and Tiedje (1984) reported that 75–100% of the theoretical yield of methane was evolved in 56 days.

6.3.2.3 Sediment and Soil

Di-*n*-butyl phthalate also rapidly degrades in soil and sediments (Staples et al. 1997). For example, 500 µg/L of di-*n*-butyl phthalate in river and estuarine sediment samples degraded to one-half the initial concentration in 1–5 days (EPA 1984a), and 92% (by $^{14}CO_2$ evolution) of the di-*n*-butyl phthalate was degraded in lake sediment in 14 days at 28 EC (Johnson et al. 1984). Inman et al. (1984) reported that 1 mg/L di-*n*-butyl phthalate in four soils was degraded to >80% (determined by $^{14}CO_2$ evolution) within 80 days in almost every case. Silica sand and a sterile control showed almost no degradation. One of the soils, Plainfield, required 200 days to reach 89.6% $^{14}CO_2$ evolution. In four samples of Little Popo Agie River (Wyoming) sediment, Heitkamp and Johnson (1984) reported that di-*n*-butyl phthalate, at a concentration of 25.6 µg/L, was degraded aerobically to 71–75% (measured by $^{14}CO_2$ evolution) in 14 days. In one sample, degradation was reported to be only 2.2%. Samples were taken from both up- and downstream of an oil field waste water discharge point; the sediment that degraded only 2.2% of the initial di-*n*-butyl phthalate was taken at the discharge point, which suggests that toxic substances may have retarded the degradation.

6.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

6.4.1 Air

As presented in Chapter 4, di-*n*-butyl phthalate has a relatively low vapor pressure and Henry's law constant, as well as a relatively high octanol/water partition coefficient and soil sorption coefficient. This combination of properties is consistent with a chemical that is found to only a limited extent in air. Nonetheless, di-*n*-butyl phthalate appears to be widespread in air at low levels, with urban air having somewhat higher concentrations than air in rural or uninhabited areas. It should be noted that because phthalates are widespread, they appear in the laboratory environment, and considerable effort must be exercised by analysts to ensure that external contamination has not influenced the results. The monitoring studies reported here appear to have taken reasonable efforts to eliminate contamination from their analyses. Di-*n*-butyl phthalate has been reported over the Pacific and Atlantic Oceans at levels of approximately $1x10^{-6}$ mg/m^3 (1 ng/m^3) (Atlas and Giam 1981; Giam et al. 1980), in outdoor air in Sweden, the United States (Thurén and Larsson 1990), and Canada (Otson et al. 1991) at $2.3x10^{-7}$–$4.99x10^{-5}$ mg/m^3 (0.23–49.9 ng/m^3), over New York City at levels of $3.3x10^{-6}$–$5.7x10^{-6}$ mg/m^3 (3.3–5.7 ng/m^3) (Bove et al. 1978), in industrialized areas along the Niagara River at levels of $4.5x10^{-6}$ mg/m^3 (4.5 ng/m^3) as vapor and at $6.2x10^{-6}$ mg/m^3 (6.2 ng/m^3) in particulate matter (Hoff and Chan 1987), in indoor air in a "sick" building at levels of $1.2x10^{-3}$–$5.9x10^{-3}$ µg/m^3 (1,200–5,900 ng/m^3) (but not in the air on the roof of the building) (Weschler et al. 1990), in residences in Canada at >$1.0x10^{-5}$ mg/m^3 (>10 ng/m^3) (Otson et al. 1991), cigarette and wood smoke (Otson et al. 1991), and waste incinerator stacks at $7.7x10^{-3}$ mg/m^3 (7,700 ng/m^3) (Jay and Stieglitz 1995; Nishikawa et al. 1992). Crump (1995) reported that 0.04 mg/m^3 of di-*n*-butyl phthalate from paint was detected in indoor air. As noted in Section 6.2.1, Cadogan and Howick (1996) reported that an indoor overall emission rate of $2.3x10^{-4}$ mg/second-m^3 at 25 EC has been calculated for all phthalate plasticizers in products such as wall coverings, flooring, upholstery, and wire insulation.

6.4.2 Water

Overall there appears to be considerable uniformity in the concentration of di-*n*-butyl phthalate in the surface waters of the United States, if locally contaminated areas are excluded. The National Drinking Water Contaminant Occurrence Database (NDOD), which contains data from ambient water samples, lists the number of detections of di-*n*-butyl phthalate in groundwater and surface water at several locations

around the United States. Di-*n*-butyl phthalate was detected in groundwater at 1 of 363 sites (0.3% of sites) with a concentration of 57 µg/L (EPA 2000a). In lakes/reservoirs, di-*n*-butyl phthalate was detected at 2 of 15 sites (13.3% of sites) with a range and average concentration of 2.5–10.7 and 6.6 µg/L, respectively (EPA 2000a). In other surface waters, di-*n*-butyl phthalate was detected at 9 of 253 sites (3.6% of sites) with a range and average concentration of 0.2–150 and 17.1 µg/L, respectively (EPA 2000a). Water samples taken along the Mississippi River at the origin of the river in Minnesota, at the junction of the Ohio River, below Memphis, Tennessee, and just below New Orleans had di-*n*-butyl phthalate concentration of 0.15, 0.14, 0.15, and 0.14 µg/L, respectively (DeLeon et al. 1986). There was no apparent effect of input from cities, industrial sources, or tributaries along the length of the river. These data suggest that transport mechanisms rather than source factors play a major role in the distribution of di-*n*-butyl phthalate. This observation is consistent with the continuous extensive wet and dry deposition from air (see Section 6.3.1). At one site, Delaware River water contained 0.6 µg/L of di-*n*-butyl phthalate that could be traced to industrial sources (Sheldon and Hites 1979); lower average concentrations were noted downstream (range, 0.1–10.4 µg/L). The difference between upstream and downstream average concentrations was not large and may reflect analytical variability. Concentrations of di-*n*-butyl phthalate in water from the Inner Harbor Navigation Canal (which connects Lake Ponchartrain to the Mississippi River near New Orleans) were 0.5–0.7 µg/L (McFall et al. 1985b).

The results of a 10-city drinking water survey reported the presence of di-*n*-butyl phthalate in 6 of 10 city water supplies. Levels ranged from 0.1 to 0.2 µg/L for five cities; the level was 5.0 µg/L for one city (Keith et al. 1976).

Bauer and Herrmann (1997) reported that di-*n*-butyl phthalate was present in the leachate from various fractions of household wastes from the regions of Bayreuth and Straubling in Germany. The wastes included food waste, paper for recycling, unusable paper, cardboard, plastic films, other plastics, textiles, 8–40 mm screened fraction, <8 mm screened fraction, compound packing waste, compound materials, and disposable diapers. Leachate from a mixture of all waste categories except food waste contained a maximum of 34 µg/L of di-*n*-butyl phthalate, while leachate from a mixture of waste categories limited to plastic films, other plastics, textiles, 8–40 mm screened fraction, <8 mm screened fraction, and compound materials contained a maximum of 63 µg/kg di-*n*-butyl phthalate. The authors were careful to exclude inadvertent sources of phthalate esters. This report indicates that di-*n*-butyl phthalate is present in European household waste and that it leaches from that waste to percolating water. The extent to which

this occurs in a landfill is unclear as is whether or not the dissolved di-*n*-butyl phthalate leaches to groundwater after leaving landfills.

6.4.3 Sediment and Soil

Considerable variability is encountered in the available data on sediments and soils. In a study of sediments from Los Angeles Sanitation District's sewage outfalls, di-*n*-butyl phthalate was reported at 5 sites with concentrations ranging from 118 to 355 µg/kg dry weight (Swartz et al. 1983). Similar values were obtained along the Detroit River. Detectable di-*n*-butyl phthalate concentrations were reported in 4 of 13 samples, with values ranging from 190 to 650 µg/kg dry weight. Marine sediment from San Luis Pass, Texas had approximately 15–93 µg/kg dry weight of di-*n*-butyl phthalate (Murray et al. 1981). Bauer and Herrmann (1997) reported that di-*n*-butyl phthalate was present in household waste in Europe (see Sections 6.4.2 and 6.4.4) and leached from waste. It is anticipated that this leaching will occur in a municipal waste landfill and that di-*n*-butyl phthalate will enter the soil surrounding these landfills. It is further anticipated that this European waste is not considerably different from waste generated in the United States.

6.4.4 Other Environmental Media

Very little recent information on di-*n*-butyl phthalate concentrations in other environmental media in the United States is available. Recent information, however, is available for some European countries and for Canada, and these studies show the presence of di-*n*-butyl phthalate in a range of foods and textiles. It should be noted that di-*n*-butyl phthalate is approved by the U.S. Food and Drug Administration (FDA) for use as an indirect food additive in adhesives (21 CFR 175.105), in paper and paperboard in contact with aqueous and fatty foods (21 CFR 176.170), in cellophane (not to exceed 5%) (21 CFR 177.1200), and in rubber articles intended for multiple use (21 CFR 177.2600), and thus, di-*n*-butyl phthalate may enter food products indirectly in the United States. Some of the European studies show migration from adhesives into foodstuffs. In addition, di-*n*-butyl phthalate is used as a lubricant in the manufacture of textiles and has been found in clothing in Europe. Mean concentrations of di-*n*-butyl phthalate in food group samples, from 1993 Total Diet Study conducted by the Ministry of Agriculture, Fisheries & Food (MAFF) in Great Britain, were 0.003, 0.09, 0.1, and 0.2 µg/g (wet weight) for milk, carcass meat, eggs, and poultry, respectively (MAFF 1996a). In a market basket survey of 98 different food types obtained from Halifax, Canada in 1986, di-*n*-butyl phthalate was detected in butter (1.5 µg/g), freshwater fish

(0.5 µg/g), cereal products (ranged from not detected up to 0.62 µg/g), baked potatoes (0.63 µg/g), coleslaw (0.11 µg/g), bananas (0.12 µg/g), blueberries (0.09 µg/g), pineapples (0.05 µg/g), margarine (0.64 µg/g), white sugar (0.2 µg/g), and gelatin dessert (0.09 µg/g) (Health Canada 1994). In Germany, the concentration of di-*n*-butyl phthalate in milk and milk products ranged from not detected (i.e., <0.01 mg/kg) to 0.07 mg/kg (Bruns-Weller and Pfordt 2000a, 2000b). The authors noted that di-*n*-butyl phthalate concentrations in milk purchased at the retail level were no higher than raw milk. The concentration of di-*n*-butyl phthalate in retail infant formula from Great Britain ranged from <0.05 to 0.09 µg/g (dry weight basis) (MAFF 1998). In Germany, levels of di-*n*-butyl phthalate in baby food ranged from not detected (i.e., <0.01 mg/kg) to 0.03 mg/kg (Bruns-Weller and Pfordt 2000a, 2000b).

Di-*n*-butyl phthalate may enter food materials by uptake from the environment. For example, reported concentrations of di-*n*-butyl phthalate in fish ranged from 78 to 200 µg/kg (Giam and Wong 1987; Stalling et al. 1973; Williams 1973). Concentrations of di-*n*-butyl phthalate in fish from various Great Lakes (United States) harbors and tributary mouths ranged from $<2x10^{-5}$ to $3.5x10^{-2}$ µg/kg wet weight (de Vault 1985). Oyster and clam concentrations of di-*n*-butyl phthalate ranged from 40 to 570 µg/kg (McFall et al. 1985a; Ray et al. 1983). Ishida et al. (1981) reported the presence of di-*n*-butyl phthalate in egg white (but not in yolk) collected from six regions of Japan. Concentrations of di-*n*-butyl phthalate ranged from 50 to 150 µg/kg.

Table 6-2 summarizes the concentration of di-*n*-butyl phthalate found in food packaged in paper and board packing materials (range, <0.02–62 µg/g food), and the concentration of di-*n*-butyl phthalate in the packaging materials used to contain the foods (range, <5–5,860 µg/g packaging material) (MAFF 1995). In another study, Aurela et al. (1999) found that di-*n*-butyl phthalate could migrate into sugar from adhesives used in the joints of paper packaging materials. After storage in a package for 4 months, the sugar contained 0.5–1.0 mg/kg of di-*n*-butyl phthalate; unpackaged sugar did not contain any phthalates. Higher concentrations were reported for that portion of the sugar that touched the seams of the packaging material. Di-*n*-butyl phthalate found in coffee filter paper may contaminate coffee beverages during the brewing process. For example, two of nine samples of coffee filters contained 8,200 and 15,000 µg/kg di-*n*-butyl phthalate (Fricker and Hardy 1990).

Table 6-3 summarizes the concentration of di-*n*-butyl phthalate detected in various categories of waste from European sources (Bauer and Hermann 1997). The authors also calculated that 11.4–106.7 mg/kg di-*n*-butyl phthalate was present in the waste on a dry-weight basis and constituted the second most

Table 6-2. Concentration of Di-*n*-butyl Phthalate in Paper and Board Packaging and Food Packaged in Paper and Board

Food types	Number of samples	Concentration in packaging (mg/kg)[a]	Number of samples	Concentration in food (mg/kg)[b]
Bakery products[c] and snacks	20	<5–2,560	5	<0.02–0.9
Confectionery	17	20–550	7	<0.02–5.8
Meat and fish products	7	40–1,380	2	0.05, 4.4
Fats	21	20–2,500	5	outer, 1.5–8.4 core, 1.4–8.7
Pasta and cereals	11	16–110	2	<0.02, 0.5
Dried fruits	2	10–16	1	<0.02
Flour and sugar	9	<5–450	2	0.2, 1.6
Gravy granules and parmesan cheese[d]	7	150–3,160	5	0.8–62
Miscellaneous	6[e]	29–5,860	2[f]	0.04, 10

Source: MAFF (1995)

[a]Limit of detection was 5 mg/kg packaging.
[b]Limit of detection was 0.02 mg/kg food.
[c]"Backey products" in original source.
[d]These products have been grouped together as they have the same type of packaging.
[e]This group consists of samples of short crust pastry mix, Yorkshire pudding, an ice lolly, waffles, eggs, and vegetable burger mix.
[f]This group consists of one sample of an ice lolly and vegetable burger mix.

Table 6-3. Concentration of Di-*n*-butyl Phthalate in Categories of Household Waste

	Concentration of di-*n*-butyl phthalate (mg/kg)[a,b]		
Waste fraction	Minimum	Maximum	Mean
Food waste	1.8	10.2	5.6
8–40 mm fraction	5.1	67.1	38.1
<8 mm fraction	10.3	17.6	15.0
Paper for recycling	9.6	22.0	15.2
Unusable paper	6.8	22.3	11.6
Cardboard	14.4	121.7	45.6
Plastic films	13.5	105.0	36.2
Other plastics	36.1	762.6	181.2
Textiles	6.0	18.9	10.9
Compound packing waste	60.1	115.3	82.4
Compound materials	609.7	2,160.0	1,473.0
Disposable diapers[c]	3.2	43.0	11.0

Source: Bauer and Herrmann (1997)

[a]Results are from six extractions except “compound materials”, for which the results are for nine extractions.
[b]The listing of the precision of the values presented in this table is the same as the original paper.
[c]Described as “nappies” in the original paper.

commonly found phthalate ester, constituting 6.0–6.7% of the total phthalates found in the waste (bis-2-ethylhexyl phthalate was the most common phthalate ester, constituting 91.9–93.3% of the total phthalates found).

Bruns-Weller and Pfordt (2000b) reported that a variety of clothing sampled in Germany contained levels of di-*n*-butyl phthalate ranging from 0.66 to 30.43 mg/kg. Di-n-butyl phthalate is used as a lubricant for textiles (see Section 5.3).

6.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

Di-*n*-butyl phthalate is present in both urban and rural air (1–6 ng/m^3), some drinking waters (0.1–0.5 µg/L; the data are >20 years old), and some foods (3–1,500 µg/kg) (see above). Thus, the general population is likely to be exposed to di-*n*-butyl phthalate at levels that are generally in the low ppb range or below. It should be noted that some of the U.S. data presented above are 10 or more years old, so current exposure levels might be different. Based on these data, the highest exposure to di-*n*-butyl phthalate is most likely to come from some dairy products, fish, and seafood, if these foods comprise a large part of the diet. Di-*n*-butyl phthalate is also an indirect food additive that is present in food containers and may be transferred to foods. There is evidence of this in European and Canadian studies, but not in U.S. studies. It may be more likely that, for the general population, air is the main source of di-*n*-butyl phthalate exposure if dairy products and seafood are not important in the diet. No data were available that could be used to estimate doses from dermal exposure to di-*n*-butyl phthalate.

Chan and Meek (1994) estimated the daily intake of di-*n*-butyl phthalate by the population of Canada (see Table 6-4). Based on the medium-specific intakes, it is estimated that the average daily intake of di-*n*-butyl phthalate for the various age groups in the general population in Canada range from 1.9 to 5.0 µg/kg body weight/day. It should be noted that these estimates do not include intake from consumer products (e.g., nail polish). Based on the percentage of di-*n*-butyl phthalate in some cosmetics (from 0.1 to between 10 and 25%), these products could contribute significantly to the exposure of some members of the general population.

The National Institute for Occupational Safety and Health (NIOSH) conducted the National Occupational Exposure Survey (NOES) from 1981 to 1983 (NIOSH 1988a). Data from the survey show that an estimated 31,502 facilities use di-*n*-butyl phthalate in 199 industries involving 138 occupations (RTECS

Table 6-4. Estimated Daily Intake of Di-*n*-butyl Phthalate by the Population of Canada

Substrate/ medium[a]	Estimated intake by age groups of di-*n*-butyl phthalate (µg/kg body weight/day)				
	0–0.5 years[b]	0.5–4 years[c]	5–11 years[d]	5–19 years[e]	20–70 years[f]
Ambient air	0.00021–0.00030	0.00033–0.00040	0.00033–0.00041	0.00028–0.00038	0.00025–0.00034
Indoor air	0.68	0.91	1.1	0.87	0.78
Drinking water	0.11	0.062	0.033	0.022	0.021
Food	1.6	4.1	3.2	1.4	1.1
Soil	<0.0005–0.0070	<0.00038–0.0054	<0.00013–0.00049	<0.000035–0.00049	<0.000028–0.00040
Total estimated intake	2.4	5.0	2.3	2.3	1.9

Source: Chan and Meek 1994

[a]Mean concentrations in ambient air based on a small study in a limited region of Ontario were 4.5–6.2 ng/m^3; the maximum concentration in indoor air was 2.85 µg/m^3 based on a small and possibly unrepresentative number (n=9) of homes in Montreal; mean values were not specified. It is assumed that people generally spend 4 hours outdoors and 20 hours indoors. Di-*n*-butyl phthalate was not detected in drinking water in a regional study in Ontario (limit of detection, 1.0 µg/L); mean values in surface water and groundwater supplies in Alberta were 1.0 µg/L. Di-*n*-butyl phthalate was detected in butter (1.5 µ/g), fresh water fish (0.5 µg/g), cereal products (ranged from not detected up to 0.62 µg/g), baked potatoes (0.63 µg/g), coleslaw (0.11 µg/g), bananas (0.12 µg/g), blueberries (0.09 µg/g), and pineapples (0.05 µg/g), margarine (0.64 µg/g), white sugar (0.2 µg/g), and gelatin dessert (0.09 µg/g). The detection limits, which were not specified for individual foodstuffs, varied depending on the reagent blank values, interferences arising from coextracted food components, and the fat content of the food (range, 0.01–0.5 µg/g); the content in food stuffs in which di-*n*-butyl phthalate was not detected was considered to be 0. Calculated intakes are based upon consumptions of individual food composites for each age group in the population. The di-*n*-butyl phthalate content in the soil in urban areas of Port Credit, Oakville, and Burlington, Ontario, ranged from <0.1 to 1.4 µg/g. Available data were insufficient to estimate intake from consumer products, though cosmetics may contribute significantly to the exposure of some members of the general population in certain age groups, based on the percentage content of some products (0.1 to between 10 and 25%).
[b]Assumed to weigh 7 kg, breathe 2 m^3 air, drink 0.75 L water, and ingest 35 mg soil.
[c]Assumed to weigh 13 kg, breathe 5 m^3 air, drink 0.8 L water, and ingest 50 mg soil.
[d]Assumed to weigh 27 kg, breathe 12 m^3 air, drink 0.9 L water, and ingest 35 mg soil.
[e]Assumed to weigh 57 kg, breathe 21 m^3 air, drink 1.3 L water, and ingest 20 mg soil.
[f]Assumed to weigh 70 kg, breathe 23 m^3 air, drink 1.5 L water, and ingest 20 mg soil.

1999). Exposed populations are estimated to be 512,631 employees, of whom 198,249 are female. Exposures in occupational settings can occur through skin contact and by inhalation of vapors and dust. Phthalates are manufactured within closed systems, but workers can be exposed during filtering or loading/unloading of tank cars (NTP 2000). Higher exposures to phthalates can occur during incorporation of the phthalate in the final product (e.g., plastics) if the process runs at a high temperature. In a limited number of surveys, di-*n*-butyl phthalate levels in production facilities in the United States have ranged from concentrations below the detection limit (0.01–0.02 mg/m^3) to 0.08 mg/m^3 (NTP 2000). Following a review of six studies, the American Chemistry Council has estimated exposure to di-*n*-butyl phthalate in the workplace based upon an assumed level of 1 mg/m^3 in the air during the production of phthalates (NTP 2000). Exposure levels during incorporation of di-*n*-butyl phthalate into plastic are not known. An exposure level was estimated by using assumptions of a 10 m^3/day inhalation and a 70 kg body weight. The resulting exposure estimate was 143 µg/kg body weight/workday for workers employed in phthalate manufacturing (NTP 2000).

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).

Studies from Europe and Canada suggest that children may be exposed to di-*n*-butyl phthalate from food. However, little information is available that would allow an estimation of exposures from food and the differences between adult and child diets from U.S. studies. Infant food from Germany had only traces of di-*n*-butyl phthalate at levels ranging from not detected (i.e., <0.01 mg/kg) to 0.03 mg/kg (Bruns-Weller and Pfordt 2000a, 2000b). Mother's milk samples from German women also contained low amounts of

di-*n*-butyl phthalate at concentrations in the range of 0.01–0.05 mg/kg. Di-*n*-butyl phthalate was detected in infant formula from Great Britain at concentrations ranging from 0.08 to 0.40 mg/kg (dry weight basis) in 1996 and <0.05–0.09 mg/kg (dry weight basis) in 1998 (MAFF 1996b, 1998). While di-*n*-butyl phthalate has been found in food in the United States, mainly fish, it is difficult to determine how widespread contamination of food is, based on the available information, some of which is >20 years old.

Chan and Meek (1994) estimated the daily intake of di-*n*-butyl phthalate by children in Canada (see Table 6-4). Based on the medium-specific intakes, it is estimated that the average daily intake of di-*n*-butyl phthalate for age groups from 0–11 years in the general population in Canada range from 2.4 to 5.0 µg/kg body weight/day. In Great Britain, the range of average exposure from infant formula was 2.4 and 1.4 µg/kg body weight/day at birth and 6 months, respectively, in 1998 (MAFF 1998). In 1996, the average exposures from infant formula were 14 and 9.3 µg/kg body weight/day at birth and 6 months, respectively.

Children, and especially small children, may be exposed to elevated levels of di-*n*-butyl phthalate from chewing on soft plastic material if it contains this chemical as a plasticizer; however, it is unclear exactly what products may contain di-*n*-butyl phthalate. Di-*n*-butyl phthalate can migrate from plastics into surrounding media, including saliva and air. For example, Lygre et al. (1993) reported that di-*n*-butyl phthalate in denture material (polymethyl methacrylate) leached into saliva in the mouths of adult patients with new dentures. The concentration range was not detected–7.5 mg/L with a mean concentration of 0.1 mg/L. Out of 11 adult patients, 7 had detectable levels of di-*n*-butyl phthalate. No di-*n*-butyl phthalate was detected in the saliva of adult patients with old dentures or in the controls (the detection limit was 0.1 mg/L). In addition, 0.04 mg/m^3 of di-*n*-butyl phthalate, which had volatilized from paint, was detected in indoor air (Crump 1995). Cadogan and Howick (1996) reported that an indoor overall emission rate of 2.3x10^{-4} mg/second-m^3 at 25 EC has been calculated for all phthalate plasticizers in products such as wall coverings, flooring, upholstery, and wire insulation. Since the physical processes associated with the leaching of plasticizers from plastics should be the same for all plastics, it is anticipated that di-*n*-butyl phthalate will also migrate from other plastics to whatever medium is present at the plastic interface. Di-*n*-butyl phthalate is used in vinyl flooring and floor wax, and children, especially young children, often crawl and play on floors. No data are available that allow a quantification of the number or types of plastic products that contain di-*n*-butyl phthalate as a plasticizer; however, latest public production figures show that approximately 6,662 metric tons (1.5x10^7 pounds) of di-*n*-butyl

phthalate were produced in the United States in 1993 (see Chapter 5). Some plastics can contain >50% plasticizer (Cadogan and Howick 1996) and, based on the presence of di-*n*-butyl phthalate in the air and water, at least some of this is likely to escape either by volatilization or other transfer process to the surrounding material (e.g., saliva, skin, water). Thus, it appears that children may be more highly exposed to di-*n*-butyl phthalate than adults because they are potentially exposed dermally while lying on floors and carpets, and by inhalation by playing near potential sources such as flooring and carpeting. Mouthing of toys is another potential source of oral phthalate exposure in children. However, use of di-*n*-butyl phthalate in toys appears to be rare. In an analysis of 17 plastic toys, di-*n*-butyl phthalate was detected in one polyvinyl chloride doll's head at 0.01% by weight (Rastogi 1998). No information, however, is available that would allow the confirmation of this possibility or the quantification of the potential exposures.

There are no specific examples of di-*n*-butyl phthalate contamination in homes resulting from inadvertent transport to the home from workplace exposures.

Young children sometimes ingest soil either intentionally through pica (the desire to eat substances, such as soil or chalk, not normally eaten) or unintentionally through hand-to-mouth activity. While no childhood exposures to di-*n*-butyl phthalate from soil ingestion or dermal soil exposure have been documented, this phthalate has been detected in soil. In addition, di-*n*-butyl phthalate has also been detected in atmospheric fallout in both urban and rural areas, and is not expected to migrate rapidly to groundwater or to volatilize rapidly from soil surfaces. Nonetheless, di-*n*-butyl phthalate is anticipated to degrade rapidly, and exposures to this chemical from other sources are anticipated to be much greater. Therefore, exposures of children to di-*n*-butyl phthalate from soil would likely be slight. No information is available on the absorption in the stomach (bioavailability) of di-*n*-butyl phthalate in soil to children.

6.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Individuals who manufacture or use plasticizers probably would have the highest potential for exposure to di-*n*-butyl phthalate. People living near chemical factories or hazardous waste sites where di-*n*-butyl phthalate is present could also have higher than average exposure. Certain cosmetics, such as nail polish, contain di-*n*-butyl phthalate, and higher exposures may result to consumers of these products. In addition, since many plastic articles contain di-*n*-butyl phthalate and many of these are used in homes and businesses, indoor concentrations may be higher than outdoor concentrations, as has been reported (see

Section 6.4.1 above). Therefore, populations that work indoors may be exposed to higher concentrations than those who work outdoors. Insufficient information is available to quantify any differences.

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 di-*n*-butyl phthalate is available. Where adequate information is not available, ATSDR, in conjunction with the National Toxicology Program (NTP), is required to assure the initiation of a program of research designed to determine the health effects (and techniques for developing methods to determine such health effects) of di-*n*-butyl phthalate.

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. Data are available on the physical and chemical properties of di-*n*-butyl phthalate (see Chapter 4), and further research in this area does not appear to be required. Nonetheless, measurements of properties such as water solubility, vapor pressure, and octanol/water partition coefficient are technically difficult because di-*n*-butyl phthalate and other phthalates are ubiquitous contaminants in the environment and are contaminates of laboratory air and surfaces, making accurate measurements difficult. Hence, a set of data that are very carefully determined are needed.

Production, Import/Export, Use, Release, and Disposal. Available data indicate that di-*n*-butyl phthalate is produced in substantial amounts at several locations in the United States, is widely used in a variety of consumer products, and is subject to regulations concerning disposal; however, no publicly available current production data are available. In addition, in the specialty plasticizer market, the amount of specific plasticizers used in various applications are not available and data may change over time. Authoritative sources of current data on production, imports, exports, specific uses, releases

to environmental media, and disposal methods were not located. Collecting such data would be valuable in estimating human exposure to di-*n*-butyl phthalate.

Environmental Fate. Biodegradation is well described in the literature, and the importance of hydrolysis reactions can be inferred from sterile controls reported with biodegradation studies. Major gaps are present in the understanding of partitioning and transport of di-*n*-butyl phthalate in water. For example, the log K_{oc} of di-*n*-butyl phthalate is reported to be 3.14 (see Chapter 4), which indicates that the majority of di-*n*-butyl phthalate in water will be sorbed to sediments or suspended particles. Many reports, however, have measured the dissolved/sorbed concentration of di-*n*-butyl phthalate in environmental samples and have found that, in some cases, in excess of 70% of the di-*n*-butyl phthalate is in the dissolved state. A greater knowledge of this process would allow a more comprehensive understanding of the exposure potential for aquatic organisms to di-*n*-butyl phthalate. Similarly, a better understanding of the partitioning of di-*n*-butyl phthalate on atmospheric particles and their washout/ fallout characteristics would also allow better exposure assessments.

Bioavailability from Environmental Media. Exposure of the general public occurs via air, water, food supply, cosmetics, and soils. However, the bioavailability of di-*n*-butyl phthalate in each of these media has not been investigated. Data of this type, especially on the availability of di-*n*-butyl phthalate that is bound to soils, sediments, and air particulates by the inhalation, oral, and dermal routes, are needed to assess the relative importance of these media to human exposure.

Food Chain Bioaccumulation. Available data indicate that di-*n*-butyl phthalate tends to be taken up and metabolized by invertebrates and fish. Numerous studies have shown that the accumulation of di-*n*-butyl phthalate in the aquatic and terrestrial food chain is limited by biotransformation, which progressively increases with trophic level (Staples et al. 1997). Therefore, di-*n*-butyl phthalate will not biomagnify through the food chain.

Exposure Levels in Environmental Media. Information on exposure levels in the environment is relatively sparse. Although a number of atmospheric air levels have been reported, it would be useful to know more specifics about urban air levels. More extensive data on food and drinking water levels of di-*n*-butyl phthalate would also be useful in assessing total human exposure.

Exposure Levels in Humans. No data are available showing the concentration of di-*n*-butyl phthalate in common plastic consumer products. A list of the composition of common plastic consumer products and the amount of di-*n*-butyl phthalate that they contain would allow a much better estimation of exposures from these products. Di-*n*-butyl phthalate has been extensively monitored in food in Europe and Canada, but very little information is available about the U.S. food supply. This type of information, especially information about infant foods and mother's milk, would permit an estimation of the importance of food in di-*n*-butyl phthalate exposures. In addition, di-*n*-butyl phthalate is an indirect food additive present in food containers, and the extent to which it migrates into food would allow a better estimation of exposures from food. While some information about migration is available from European studies, information on migration from packaging meeting U.S. standards would allow better assessments of exposures from food. In addition, few data are available on human tissue levels of di-*n*-butyl phthalate, so it is not possible at this time to assess the total impact of di-*n*-butyl phthalate on the human population. Moreover, information relating exposure levels to levels in humans are needed to assess risks to populations surrounding hazardous waste sites.

Exposures of Children. A number of studies are available that report di-*n*-butyl phthalate concentrations in infant foods and mother's milk in Europe. Similar studies are not available for U.S. sources, and this information could be used to better estimate the importance of food as a source of di-*n*-butyl phthalate exposures to children. No data are available that allow the estimation of exposures resulting from mouthing and/or chewing on plastics or other compounds containing di-*n*-butyl phthalate. It is possible that the majority of plastic toys used by children contain plasticizers, and the extent to which these contain di-*n*-butyl phthalate is not known. A complete study of the migration of di-*n*-butyl phthalate from plastics and other compounds into saliva, including the effects of chewing, also has not been performed. This information would allow a much better estimation of children's exposure to di-*n*-butyl phthalate. It is known that di-*n*-butyl phthalate migrates from polymethyl methacrylate denture material into saliva (see Lygre et al. 1993); however, dentures are present in the mouth for long periods, and the applicability of this study to di-*n*-butyl phthalate migration from plastic and other materials to saliva in children's mouths is unclear. In addition to the potential migration of di-*n*-butyl phthalate from plastic and other materials to saliva, the potential also exists for dermal and inhalation exposures. Di-*n*-butyl phthalate is also used in cosmetics, such as nail polish, and children may play with some of these products. The extent to which these products contain di-*n*-butyl phthalate is not known. While it is much more likely that exposures will occur from plastics and other materials, no data are available on childhood exposures to di-*n*-butyl phthalate from soil either from ingestion or from dermal exposure.

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 di-*n*-butyl phthalate were located. This compound is not currently one of the compounds for which a subregistry has been established in the National Exposure Registry. The compound will be considered in the future when chemical selection is made for subregistries to be established. The information that is amassed in the National Exposure Registry facilitates the epidemiological research needed to assess adverse health outcomes that may be related to the exposure to this compound.

6.8.2 Ongoing Studies

The Federal Research in Progress (FEDRIP 2000) database provides information on ongoing studies; however, none were found that fill any of the data needs identified in Section 6.8.1.

No information was located regarding ongoing studies on the environmental fate and transport of di-*n*-butyl phthalate or on levels of human exposure to di-*n*-butyl phthalate.