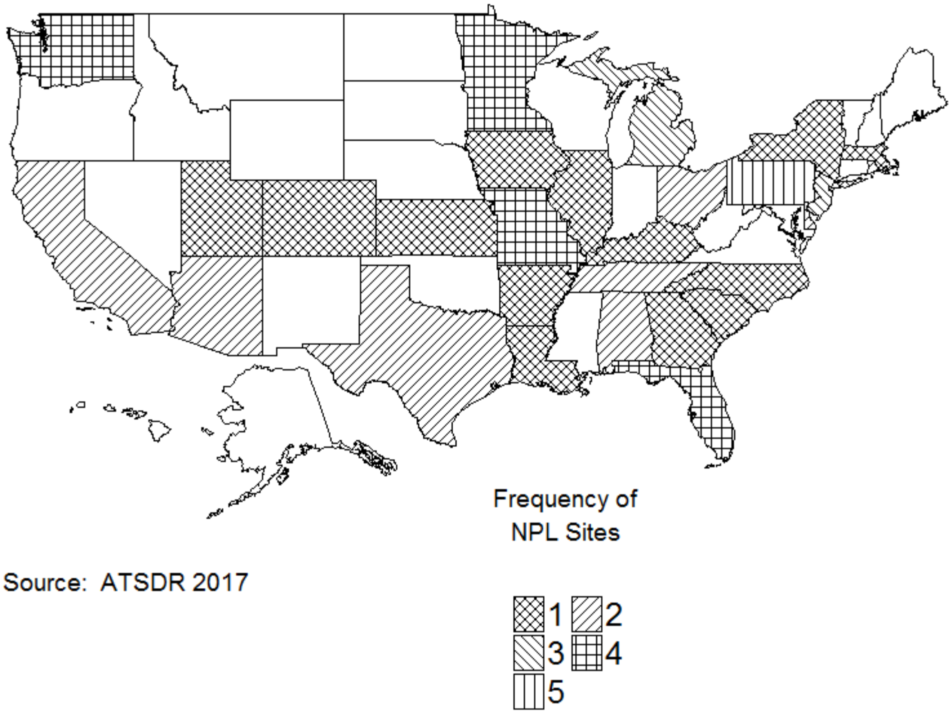
CHAPTER 5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

4,6-Dinitro-o-cresol has been identified in at least 56 of the 1,854 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2017). However, the number of sites in which 4,6-dinitro-o-cresol has been evaluated is not known. The number of sites in each state is shown in Figure 5-1. Of these sites, 56 are located within the United States.

Figure 5-1. Number of NPL Sites with 4,6-Dinitro-o-cresol Contamination



Source: ATSDR 2017

- The most likely route of exposure to DNOC for the general population is via ingestion of DNOC in drinking water.
- A daily intake of DNOC from drinking water has not been estimated, based on a lack of pertinent data.
- DNOC is primarily used as an intermediate in pesticide manufacturing.
- DNOC is not likely to volatilize appreciably from water or soil. Available data indicate that biodegradation of DNOC does not occur rapidly.

5.2 PRODUCTION, IMPORT/EXPORT, USE, AND DISPOSAL

5.2.1 Production

Table 5-1 summarizes information on U.S. companies that reported the manufacture or use of DNOC in 2016 (TRI16 2017). Toxics Release Inventory (TRI) data should be used with caution since only certain types of industrial facilities are required to report. This is not an exhaustive list.

Table 5-1. Facilities that Produce, Process, or Use 4,6-Dinitro-*o*-Cresol (DNOC)

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
IN	1	100	999	12

[a]Post office state abbreviations used.
[b]Amounts on site reported by facilities in each state.
[c]Activities/Uses:

1. Produce
2. Import
3. Used Processing
4. Sale/Distribution
5. Byproduct
6. Reactant
7. Formulation Component
8. Article Component
9. Repackaging
10. Chemical Processing Aid
11. Manufacture Aid
12. Ancillary
13. Manufacture Impurity
14. Process Impurity

Source: TRI16 2017 (Data are from 2016)

DNOC is prepared by sulfonating o-cresol with excess sulfuric acid at 80–100 °C and subsequently nitrating 4,6-disulfonic-*o*-cresol (produced by the sulfuric acid) with nitric acid or nitrous fumes (Harvey 1953). 2,6-Dinitro-*p*-cresol is prepared by nitrating *p*-cresol with nitric acid in acetic acid or a nitric acid-sulfuric acid mixture (Harvey 1953). Neither 4,6-dinitro-*m*-cresol nor 3,5-dinitro-*o*-cresol can be produced on a commercial level by the simple nitration of *o*- or *m*-cresol (Harvey 1953). The current production volume of this compound is not known, but the production volume was between 0.1 and 1.0 million pounds in 1977 (EPA 1988a).

5.2.2 Import/Export

Comprehensive current data on the import/export of the dinitrocresols were not located in the literature. However, three U.S. companies imported small amounts (<100,000 pounds or 45,300 kg) of DNOC in 1977 (EPA 1988a). Two tariff categories are defined covering a variety of meta- and ortho- forms of dinitrocresols. During 1992, tariffs were collected from imports totaling 10,719 kg of dinitrocresols;

during 1993, tariffs were collected from imports totaling 800 kg of dinitrocresols (NTDB 1994). The available information suggests a decrease in the import volumes since the late 1970s.

5.2.3 Use

DNOC is a nonsystemic stomach poison and contact insecticide. In the United States, the EPA canceled its registration as a pesticide agent starting in 1991 (EPA 1993b; Farm Chemicals Handbook 1993; HSDB 1994). It is strongly phytotoxic for broad-leaved plants, and its use as an insecticide in the United States has been limited to dormant sprays, especially for such fruit trees as apples or peaches. As a contact herbicide, it was used to control broad-leaved weeds in cereals and to desiccate potato and leguminous seed crops before harvesting (Worthing 1987). 2,4-Dinitro-6-sec-butylphenol, which is less expensive and a more effective herbicide, had begun to replace DNOC by the late 1980s (EPA 1988a). DNOC has been used as a free radical polymerization inhibitor (EPA 1988a). 2,6-Dinitro-p-cresol is used as an intermediate for synthesis of fungicides and biologically active compounds, dyes, and pharmaceuticals, and as a polymerization inhibitor for vinyl aromatic compounds (EPA 1988a; Hawley 1981).

5.2.4 Disposal

Rotary kiln incineration at a temperature range of 820–1,000°C and residence times of seconds for liquid and gaseous wastes and hours for solids can totally destroy dinitrocresols. Fluidized bed incineration at a temperature range of 450–980°C and residence times of seconds for liquid and gaseous wastes and longer for solid wastes can also destroy dinitrocresols. Mixing dinitrocresols with a more flammable solvent may facilitate incineration. Containers used for dinitrocresols that are not to be reused can be disposed by burial in a designated landfill (HSDB 1994).

5.3 RELEASES TO THE ENVIRONMENT

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

5.3.1 Air

Estimated releases of 1 pound (<0.001 metric tons) of DNOC to the atmosphere from one domestic manufacturing and processing facility in 2016, accounted for about 100% of the estimated total environmental releases from facilities required to report to the TRI (TRI16 2017). These releases are summarized in Table 5-2.

Table 5-2. Releases to the Environment from Facilities that Produce, Process, or Use 4,6-Dinitro-*o*-Cresol (DNOC)[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
IN	1	1	0	0	0	No data	1	No data	1
Total	1	1	0	0	0	0	1	0	1

[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: TRI16 2017 (Data are from 2016)

5.3.2 Water

There were no estimated releases of DNOC to surface water from one domestic manufacturing and processing facility in 2016 required to report to the TRI (TRI16 2017). There were no data on releases to publicly owned treatment works (POTWs) (TRI16 2017; see Table 5-2).

5.3.3 Soil

There were no estimated releases of DNOC to soils from one domestic manufacturing and processing facility in 2016 required to report to the TRI (TRI16 2017). There were no releases via underground injection (TRI16 2017; see Table 5-2).

5.4 ENVIRONMENTAL FATE

5.4.1 Transport and Partitioning

In laboratory experiments, photolysis of *o*-cresol in the presence of nitrogen oxides produced dinitrocresols in the aerosol phase (Grosjean 1984, 1985). It was, therefore, presumed that dinitrocresols would be present in the ambient atmosphere in aerosol form (Grosjean 1984, 1985). The distance of atmospheric transport for DNOC depends on the half-life and the physical state of the compound in air. Residence times during atmospheric transport can be sufficiently long so that such physical removal processes as wet or dry deposition may be important. The efficiency of both wet and dry precipitation is higher for particulate matter than for compounds that exist in the gas phase in the air (Schroeder et al. 1987). Therefore, atmospheric dinitrocresols, which exist predominantly in the particulate phase, may be removed by rain and snow, and these compounds may not be transported long distances from their source of emission. The detection of DNOC in rain and snow, and the observation that the ratio of concentrations of DNOC in rainwater to concentrations in air during a rain event was 5.6×10^4, confirm the importance of these removal processes (Alber et al. 1989; Leuenberger et al. 1988; Tremp et al. 1986). Precipitation of atmospheric dinitrocresols transports the compound from air to land and water.

The pK_a value of 4.4 (see Table 4-2) for DNOC suggests that in natural waters with a pH 5–9, >50% of the compound exists in the ionic state at pH 5 and the percent of ionic forms increases as the pH increases. In addition to this dissociation effect, DNOC may form H-bonds in water (EPA 1979), reducing its vapor pressure and chances of volatility from water. Using a Henry's law constant value of 1.4×10^{-6} atm-m^3/mole (Shen et al. 1982a, 1982b) and an estimation method (Thomas 1990), the estimated volatilization half-life of DNOC from a typical river 1 m deep, with a current speed of 1 m/second, and an overhead wind speed of 3 m/second, is 36 days. Therefore, direct volatilization from water will not be significant for DNOC.

The partitioning of DNOC from water to solids present in water transports the compound from the water phase to suspended solids and sediment. The adsorption of DNOC from water by suspended solids and sediment is pH dependent, and the adsorption increases as the pH of the solution decreases (Frissel and Bolt 1962; Jafvert 1990). The adsorption of DNOC also depends on the clay and organic carbon content of the suspended solids and sediment; an increase in either value increases adsorption (EPA 1979; Frissel and Bolt 1962; Jafvert 1990). This adsorption will decrease the concentration of DNOC in water. DNOC's adsorption coefficient (a Freudlich sorption parameter, Kp) of 590 mg/g (Dobbs et al. 1987; Dobbs et al. 1989) indicates that the compound moderately sorbs to suspended solids and sediment in water. However, in Rhine River water with a pH of 7.9 (Wanner et al. 1989), only an estimated 9.3% of DNOC accumulated in bottom sediment (Halfon and Bruggemann 1989). This low adsorption may be due to high water pH, lack of clay, or a low organic carbon content of the sediment, or a combination of these factors.

No experimental data regarding the bioconcentration potential of DNOC in aquatic organisms were located. Based on an estimated bioconcentration factor (BCF) of 40 (Kenaga 1980), the bioconcentration of DNOC in aquatic organisms may not be significant; however, based on an estimated log octanol/water partition coefficient (log K_{ow}) value of 2.85, DNOC may bioaccumulate in aquatic organisms (Loehr and Krishnamoorthy 1988). Given that DNOC exists predominantly in ionic forms in most natural waters (pH 5–9) and that the compound is markedly toxic to fish, bioconcentration is not expected to be important (EPA 1979).

Given the low values for vapor pressure (1.05×10^{-4} mm Hg) (see Table 4-2) and Henry's law constant (1.4×10^{-6} atm·m^3/mol) (see Table 4-2), and the consideration that the majority of the compound will be either in an ionic state or tied up through H-bonds, volatilization is not a significant transport process for DNOC from soil to the air. However, some loss of DNOC by volatilization via co-distillation with water may occur, as observed (Kaufman 1976) in the case of dinoseb, with its active ingredient 2,3-dinitro-6-sec-butyl-phenol. Volatilization is expected to occur more readily with an increase in soil acidity (which facilitates the formation of undissociated species DNOC), moisture content, and temperature (Kaufman 1976); however, a laboratory study of two types of soil found no loss of DNOC by volatilization in 65 days (Loehr 1989). The adsorption of DNOC to soil increases with a decrease in soil pH and an increase in clay and organic carbon contents of soil. The estimated soil sorption (K_{oc}) values ranging from 2.35 to 2.77 indicate that this compound is moderately adsorbed in soil. Therefore, DNOC shows moderate mobility in soil, but because of its short persistence in soil (<1 month), the compound may not leach beyond 5 cm (Ammon 1985). Similar conclusions were reached by other investigators. In spite of moderate mobility, DNOC leaching was not observed from soil columns with at least 16 bed volumes of leachate (Pohland et al. 1987). On the other hand, the water soluble salts of DNOC (sodium, potassium, calcium, and ammonium) might be expected to leach into soil. Although no experimental data were located, it seems likely that DNOC will transfer to adjacent surface water or land via runoff water from treated fields or waste sites.

5.4.2 Transformation and Degradation

Air. The two processes likely to remove dinitrocresols from the atmosphere are reactions with hydroxyl and nitrate radicals (Atkinson et al. 1992). No experimental kinetic data are available for these two reactions (Grosjean 1991). The rate constant for the gas phase reaction of dinitrocresols with OH radicals is 3.0×10^{-14} cm^3/molecule-second (Grosjean 1991). Using the method of Atkinson (1988), the estimated rate constant for this reaction is 2.1×10^{-13} cm^3/molecule-second. Based on an average ambient atmospheric concentration of OH radicals in the northern hemisphere of 5×10^5 radicals/cm^3 (Atkinson 1988) and either of the rate constant values, the estimated half-life of the DNOC reaction with OH radicals is >77 days. Since dinitrocresols are expected to be present predominantly in the particulate phase in the atmosphere, the reaction rate will be even slower compared to the gas phase reaction rate (Grosjean 1991). The reactions of phenol and cresols with NO_3 radicals may be significant processes in the air (Atkinson et al. 1992). However, the products of these reactions with phenol or cresols are *o*- and possibly *p*- substituted nitrophenol and cresol compounds (Atkinson et al. 1992). Since both *o* and *p* positions are already occupied by nitro substituents, the reaction of DNOC with NO_3 radicals does not seem to be a significant atmospheric process.

Photolysis of dinitrocresols is another reaction that can be significant for the destruction of these compounds in the air. In water, the neutral DNOC species has a light absorption spectrum with a shoulder at 305 nm (Schwarzenbach et al. 1988). Therefore, it is possible that atmospheric DNOC will absorb sunlight and undergo a reaction such as nucleophilic displacement of the nitro group by a hydroxyl group. Experimental evidence of such transformation reactions is not available (Kaufman 1976). Photolysis of a structurally similar compound, dinoseb (which has a sec-butyl group in place of the *o*-methyl group), produced side-chain hydroxylation or unsaturation (of the sec-butyl group), but no nucleophilic substitution of nitro groups (Kaufman 1976).

Water. Both neutral and anionic species of DNOC show absorption shoulders at wavelengths >300 nm (Schwarzenbach et al. 1988). However, photolysis of DNOC in water involving nucleophilic displacement of the nitro group by the hydroxyl group does not seem likely (EPA 1979). The photochemical reduction of the nitro group in DNOC is possible in water in the presence of a reducing agent (e.g., ascorbic acid or ferrous ions) and a sensitizer, such as chlorophyll (EPA 1979). However, there is no experimental evidence of the photochemical reduction of DNOC in water.

The estimated rate constants for the reaction of DNOC and 2,6-dinitro-*p*-cresol with singlet oxygen in water at pH ~7 are 1.25×10^5/molecule-second and 1.43×10^7/molecule-second, respectively (Tratnyek and Holgne 1991). Based on an average concentration of singlet oxygen in eutrophic freshwater of 4×10^{-14} M (Tratnyek and Holgne 1991) and the above reaction rate constants, this reaction may be insignificant for DNOC. The estimated half-life for 2,6-dinitro-*p*-cresol due to this reaction is 14 days, and it may be a significant process for the destruction of 2,6-dinitro-*p*-cresol in eutrophic freshwater.

Several pure cultures of microorganisms isolated from soil or sediment, such as *Corynebacterium simplex* (Gundersen and Jensen 1956; Jensen and Gundersen 1955), *Rhizobium leguminosarum* (Hamdi and Tewfik 1970), *Veillonella alkalescens* (McCormick et al. 1976), unadapted or phenol-adapted *Pseudomonas* sp. (Chambers and Kabler 1964; Tewfik and Evans 1966), and *Azotobacter* sp. (Wallnoefer et al. 1978), can biodegrade DNOC. Above a certain concentration, DNOC may be toxic to organisms. For example, at concentrations >500 mg/L, DNOC may be toxic to *C. simplex* (Bollen 1961). The degradation pathway will depend on the microorganism. It has been shown that *C. simplex* releases nitro groups of DNOC as nitrite ions (Gundersen and Jensen 1956). *Pseudomonas* sp. may biodegrade DNOC by ring cleavage. Successive replacement of nitro groups with hydroxyl groups can also occur, forming trihydroxytoluene (Golovleva et al. 1992; Tewfik and Evans 1966). The biodegradation may proceed by the successive reduction of nitro groups to amino groups by *V. alkalescens* and a *Pseudomonas* sp. (McCormick et al. 1976; Williams 1977). The metabolites that have been isolated as biodegradation products are 6-amino-4-nitro-*o*-cresol, 6-acetamido-4-nitro-*o*-cresol, 2-methyl-6-nitro-catechol, 2-methyl-6-amino-catechol, and 2,3,5-trihydroxytoluene (Tewfik and Evans 1966; Wallnoefer et al. 1978). Although these studies with pure cultures of microorganisms are important to establish degradative pathways, their relevance to environmental situations is uncertain.

The biodegradation of DNOC was also tested with a mixture of microorganisms in activated sludge (Thorn and Agg 1975; Zahn and Wellens 1980), garden soil, compost, river mud, sediment of a waste lagoon (Tabak et al. 1964), and settled domestic waste water (Tabak et al. 1981). These biodegradation studies with mixtures of microorganisms concluded that DNOC does not rapidly degrade under these conditions.

A patented waste treatment process that used activated sludge with added powdered activated carbon removed 99% of DNOC from influent that contained 11 μg/L of the compound (Patterson and Kodukala 1981). However, it is difficult to separate the contribution of the biological process from the adsorption effect of the activated carbon in removing DNOC from the influent. DNOC was resistant to anaerobic biodegradation under methanogenic conditions (O'Connor and Young 1989). Both laboratory die-away tests and experiments with natural marine plankton communities showed that DNOC was resistant to anaerobic biodegradation (Kuiper and Hanstveit 1988). Based on observations following a pesticide spill on the Rhine River involving DNOC, it was estimated that half of the initial DNOC had disappeared (due to a combination of biotic and abiotic processes) within 30 days (Capel et al. 1988).

Sediment and Soil. The transformation and degradation pathways of DNOC in soil and sediment have not been studied thoroughly. The photolysis of DNOC in soil below the surface layer and in sediment is not significant due to the lack of available sunlight. DNOC does not contain any functional groups amenable to hydrolysis (EPA 1988a). It has been speculated that adsorbed DNOC may undergo hydrolysis on clay surfaces under acidic conditions (EPA 1979), but there is no experimental evidence. Biodegradation may be the most significant process for the transformation and degradation of DNOC in soil.

Work reported in Bruinsma (1960) documents that above certain dosage levels, DNOC may be toxic to many types of soil microorganisms. These findings help explain a pattern in the available literature where the biodegradability of DNOC is widely taken for granted, but where the results of different empirical studies on the persistence of DNOC in soils may show widely differing results. The effects of DNOC toxicity to soil flora also make it difficult to interpret persistent (or disappearance) findings in terms of chemical kinetics and such concepts as half-lives.

In a soil column experiment, the estimated time for degradation of one-half the original amount of DNOC was 14 days (Kincannon and Lin 1985). These results are in line with findings from field plot analyses in Germany (Hurle and Rademacher 1970), where the disappearance of one-half the initial DNOC levels took 15 days. Other investigators have estimated that the persistence of DNOC in soil ranges from 14 days to >l month (Ammon 1985; Goring et al. 1975; Jensen and Lautrup-Larsen 1967). However, in a study to determine the treatability potential of waste sludge from explosives production, no loss of DNOC was observed in two soil samples in 65 days (Loehr 1989). The soil in this experiment was not previously exposed to industrial chemicals, wastes, or any pretreatment to acclimate the microorganisms to the chemicals.

5.5 LEVELS IN THE ENVIRONMENT

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

Table 5-3 shows the lowest limit of detections that are achieved by analytical analysis in environmental media.

Table 5-3. Lowest Limit of Detection Based on Standards[a]

Media	Detection limit	Reference
Air	0.07 mg/m^3	NIOSH 1984
Drinking water	0.009 μg/L	Di Corcia and Marchetti 1992
Surface water and groundwater	0.07 μg/L	Buchholz and Pawliszyn 1993
Soil	0.005 mg/kg	Roseboom et al. 1981
Sediment	160 μg/L	EPA 1986a
Whole blood	<0.5 mg/L	Parker 1949

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

Detections of dinitrocresols in air, water, and soil at NPL sites are summarized in Table 5-4.

Table 5-4. Dinitrocresols Levels in Water, Soil, and Air of National Priorities List (NPL) Sites

Medium	Median[a]	Geometric mean[a]	Geometric standard deviation[a]	Number of quantitative measurements	NPL sites
Water (ppb)	10	44.3	55,100	5	5
Soil (ppb)	270,000	77,100	9,040	7	6
Air (ppbv)			No data		

[a]Concentrations found in ATSDR site documents from 1981 to 2017 for 1,854 NPL sites (ATSDR 2017). Maximum concentrations were abstracted for types of environmental media for which exposure is likely. Pathways do not necessarily involve exposure or levels of concern.

5.5.1 Air

Other than in workplace air (see Section 5.6), no data regarding the concentrations of DNOC in ambient air in the United States were located. From the concentration of DNOC in rainwater during a rain event in Dubendorf, Switzerland, the concentration of DNOC in air was estimated to be (from rainwater to air partition ratio) 0.05 μg/m^3 (0.06 ppb) (Leuenberger et al. 1988).

5.5.2 Water

DNOC was detected at a concentration range of 8–400 μg/L in waste water resulting from the production and purification of trinitrotoluene (Spanggord et al. 1982a). DNOC was also qualitatively detected in the waste waters of a plant in England that produced pest control chemicals (EPA 1988a) and in two effluents from unspecified chemical plants in the United States (EPA 1976b). In a project that monitored pollutant levels in urban runoff water of 1.5 cities in the United States, DNOC was not detected (detection limit unspecified) in any runoff water (Cole et al. 1984). The Great Lakes Water Quality Board has not viewed DNOC as a toxic substance of critical concern based on levels typically encountered in water from Lakes Erie and Michigan (Great Lakes Water Quality Board 1983). DNOC was detected at a concentration of <10 μg/L in water from the Potomac River near Quantico, Virginia (Hall et al. 1987). Following an accidental spill in 1986, the estimated concentration of DNOC in Rhine River water in Nauf, Switzerland, was 100–430 μg/L (Capel et al. 1988). In California, where DNOC had been used as a pesticide, DNOC was detected in five groundwater samples at a maximum concentration of 35 μg/L (Cohen 1986; Holden 1986). In 1985, DNOC was detected in rainwater from Dubendorf, Switzerland, at concentrations ranging from 0.95 to 2.9 μg/L (Leuenberger et al. 1988; Tremp et al. 1986).

5.5.3 Sediment and Soil

DNOC is expected to be found in soil near plants where the pesticide is produced and formulated, near disposal sites, and in agricultural and waste lands to which the pesticide was applied. No quantitative data regarding the levels of DNOC in soil were located. Similarly, DNOC is expected to be found in the sludge of waste treatment plants, such as pesticide manufacturing plants, trinitrotoluene production plants, and in the sediment of rivers where the pesticide has been discharged from manufacturing plants or carried by runoff water from treated lands or waste disposal sites. However, no quantitative data regarding the levels of DNOC in sludge were located. Studies on the lower Grand Calumet River around the Indian Harbor area (Hoke et al. 1993) has documented sediment concentration of DNOC ranging from 0.24 to 2.08 mg/kg (dry weight).

5.5.4 Other Media

DNOC was not found in fish collected between 1980 and 1981 from Great Lakes harbors and tributaries (Devault 1985). DNOC was detected below the tolerance level on Rumanian plums at harvest time and in potatoes from treated fields in what was formerly East Germany (HSDB 1994).

5.6 GENERAL POPULATION EXPOSURE

The general population could be exposed to DNOC from inhaling air or ingesting food and drinking water. To estimate the daily intake of DNOC by the general population from inhaling ambient air or ingesting drinking water and food, the levels of DNOC in these media must be known, and these values were not located in the literature. There was no indication in the literature that DNOC is used in any consumer products that could lead to dermal exposure.

Workers involved in manufacturing and formulating, incinerating, or spraying the pesticide on agricultural products and waste lands, and possibly workers involved in remediating Superfund sites containing this pesticide could have been or might be occupationally exposed to DNOC. Of all the possible exposure scenarios, the level of dermal exposure of workers during spraying of DNOC in the field has actually been measured. During spray-thinning of apples with liquid sprays, the estimated average dermal exposure may range from 22.5 to 63.2 mg/hour, and the corresponding average inhalation exposure may range from <0.05 to 0.4 mg/hour (Batchelor et al. 1956; Durham and Wolfe 1962; Wolfe 1976). The DNOC levels in the urine of sprayers before, during, and after the exposure period were also determined, and DNOC was detected in 5 of 183 spray operators. The DNOC concentrations in urine in these 5 samples ranged from 0.6 to 1.3 mg/L, with an average of 0.8 mg/L (Batchelor et al. 1956). The concentration of DNOC in plasma of spray operators following a total exposure period of 5–48 hours ranged from <l to 4.3 mg/kg (Batchelor et al. 1956).

5.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

As discussed (see Section 5.6), spray operators are one group of people that could have experienced potentially high exposures (higher than background levels) to DNOC. Other occupational groups discussed in Section 5.6 have the potential to be exposed to DNOC at higher levels than the general population, but no experimental evidence of higher exposure among these occupational groups was located.

Within the general population, people who live near incinerators burning DNOC, DNOC disposal facilities, and DNOC manufacturing and formulating plants are potentially exposed to higher than background concentrations of DNOC. However, no study located in the literature provided evidence of higher than background exposure to DNOC among these groups of the population. Moreover, no study demonstrated the potential for higher than background exposure to DNOC from consuming excessive amounts of certain foods (e.g., sprayed apples or contaminated fish).