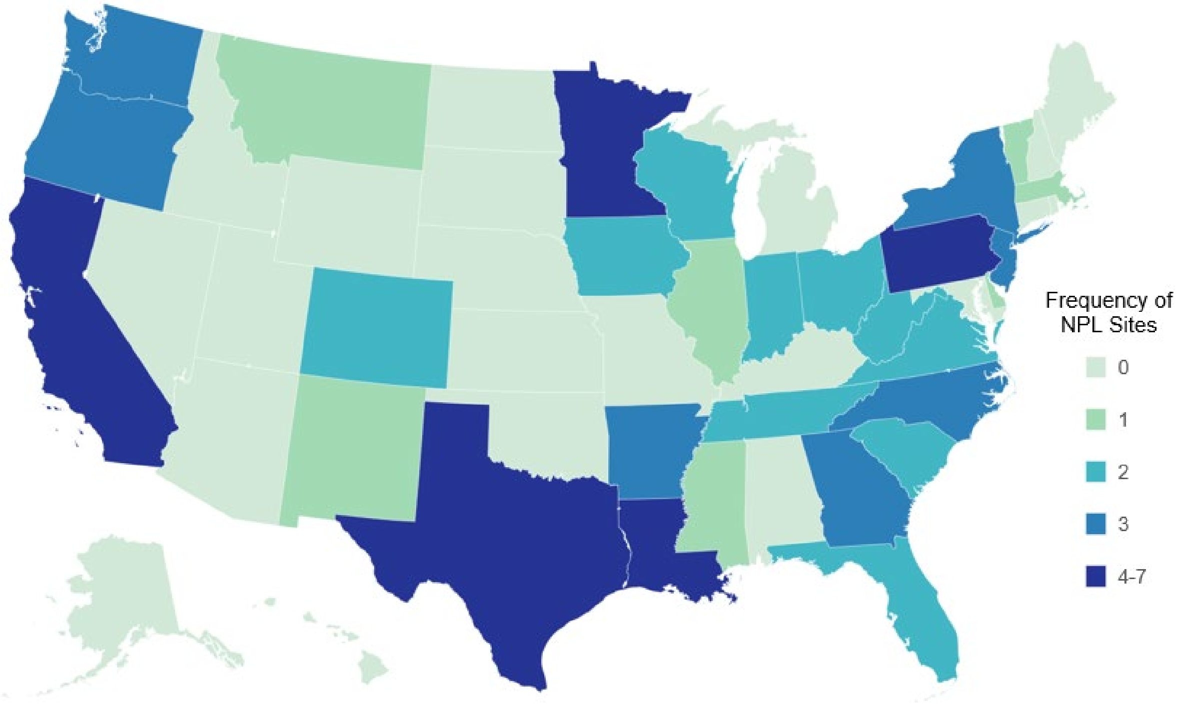
CHAPTER 5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

Coal tar creosote, coal tars, and coal tar pitch have been identified in at least 72 of the 1,868 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2022). However, the number of sites in which coal tar creosote, coal tars, and coal tar pitch has been evaluated is not known. The number of sites in each state is shown in Figure 5-1. Wood creosotes have not been evaluated at NPL sites (ATSDR 2022).

Figure 5-1. Number of NPL Sites with Coal Tar Creosote, Coal Tars, and Coal Tar Pitch Contamination



Source: ATSDR 2002

- Persons employed in industries that use coal tar creosote such as wood treatment facilities or production facilities that manufacture coal tar may be exposed to the constituents of these complex mixtures through dermal and inhalation routes. Persons using coal tar-based asphalt sealants may also be exposed to the constituents of these mixtures through dermal and inhalation pathways, and there is evidence that coal tar-based sealants may lead to increased levels of PAHs in settled indoor house dust. Family members of workers in industries using these products could be potentially exposed from contaminated work clothing or footwear.

- Dilute solutions of coal tar are used as a treatment for a variety of skin conditions, so dermal exposure from these shampoos or lotions can occur for persons using them. Wood creosote was formerly used for medicinal purposes as an expectorant, anti-septic, astringent, anesthetic, and laxative; however, today, these uses seem rare in the United States.

- Coal tar creosote is a restricted use pesticide, so it is not available to the public for wood treatment uses. In locations where accidental spills occurred or creosote was released in effluents, nearby populations may be exposed to the constituents of coal tar creosote from contaminated environmental media such as air, soil, or water. Several of the constituents of coal tar and coal tar creosote bioconcentrate in fish and aquatic organisms; therefore, ingestion of fish near contaminated sites may result in exposure to populations consuming fish in these areas.

- The fate and transport of the components of these complex mixtures will be reflective of their individual properties. In general, high molecular weight PAHs are relatively nonvolatile and are slow to biodegrade in the environment, particularly under anaerobic conditions.

- If released to water, adsorption to suspended solids will attenuate volatilization for most of the components of these mixtures, and sediment is considered an environmental sink.

- The lower molecular weight constituents are volatile and undergo oxidation in air by vapor-phase reaction with atmospheric oxidants with half-lives of a few hours to a few days.

Coal tar creosote is a complex commercial mixture of thousands of organic constituents. The most common forms are derived from coal tar distillation, yielding coal tar creosote in temperature ranges between 210 and 280°C. Coal tar and coal tar pitch share many of the PAH components of coal tar creosote. For the coal tar derivatives, the composition of the mixture varies from batch to batch depending on the coking process used (Brown et al. 2006; Gallacher et al. 2017a, 2017b). For example, Brown et al. (2006) studied 10 coal tars obtained from MGPs in the eastern United States and while there were similarities in chemical distributions of PAHs, they also noted a very wide range of bulk and chemical properties, which reflects the variability in the full chemical composition of these mixtures. Coal tar, coal tar creosote, and coal tar pitch consist primarily of PAHs and, therefore, the fate of many of the components of the mixture is similar to that of PAHs; however, the variability in final composition of these complex mixtures will impact the overall fate and transport.

Coal tar creosote has been widely used as a wood-treatment pesticide since the turn of the 20th century. As a result of this widespread and long-term use, workers in the wood-preserving industry have been exposed to coal tar creosote for many years. Human exposure to coal tar creosote can occur by inhalation or direct dermal contact. Studies have indicated that dermal exposure to creosote used in wood treatment or in coking oven processes contributed more significantly to the total body burden than respiratory exposures (Klingner and McCorkle 1994; Malkin et al. 1996; Van Rooij et al. 1993b). In other industries, such as rubber processing, occupational exposure to coal tar pitch volatiles may lead to excessive respiratory exposure to PAHs, including benzo[a]pyrene (Rogaczewska and Ligocka 1994). Individuals working in wood-preserving facilities are one of the largest exposed groups. Exposure may also occur during handling and installation of treated wood products in structures such as bridges, piers, retaining walls, cross ties, and fencing; as a result of burning treated scrap wood; and through contact with contaminated media at hazardous waste sites. In addition to PAHs, workers in coal tar and creosote industries may also be exposed to many other potentially hazardous compounds such as asbestos, silica, sulfur-substituted hydrocarbons, solvents, aliphatic amines, and aldehydes, making for a complex risk characterization (IARC 2012a). The public is unlikely to experience any significant exposure to liquid creosote through the direct use of wood preservative products because EPA canceled all non-wood uses of the material and restricted use of coal tar creosote products to certified applicators in January 1986 (EPA 1986a).

Children are exposed to the components of creosote via the same routes that adults are, but small children are more likely than adults to be in close contact with yard dirt or playground dirt, lawns, and indoor (carpet) dust, all of which may be contaminated with creosote residues. In addition, creosote residues are found in coal tar sealants for driveways, which are commonly used in the United States. Because of a tendency to put their unwashed hands and foreign objects into their mouths, and to chew on objects, children may be exposed to creosote through oral ingestion. Dermal exposure may occur through contact with treated wood used for utility poles, bridges, fences, and railroad crossties. Children may be exposed by playing near pools of discarded creosote or by playing at abandoned hazardous waste sites.

Pharmaceutical creosote preparations are derived from the processing of such woody plants as beechwood (von Burg and Stout 1992). Wood creosote (beechwood creosote) is a yellow, transparent liquid with a characteristic smoky odor, obtained by fractional distillation of wood tar. It is composed primarily of phenol, phenols, cresols, guaiacols, xylenols, and small amounts of alkyl-2-hydroxy-2-cyclopenten-1-ones. Wood creosote has been used as an expectorant, a gastric sedative, a gastrointestinal antiseptic, and particularly as an antidiarrheal agent (Ogata et al. 1993). Its current use in the United States is likely to be low or nonexistent; however, there appear to be web-based suppliers of wood creosote.

5.2 PRODUCTION, IMPORT/EXPORT, USE, AND DISPOSAL

5.2.1 Production

Table 5-1 summarizes information on companies that reported the production, import, or use of coal tar creosote for the Toxics Release Inventory (TRI) in 2022 (TRI22 2024). TRI data should be used with caution since only certain types of industrial facilities are required to report. This is not an exhaustive list. Facilities report 16 EPA criteria PAHs thought to be contained in creosote in their threshold calculations and this is what the TRI data reflect.

Table 5-1. Facilities that Produce, Process, or Use Creosote

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
AL	5	10,000	9,999,999	8
AR	4	10,000	9,999,999	8, 9, 12
CA	1	1,000	9,999	12
CT	1	100,000	999,999	8
DE	1	10,000	99,999	12
IL	2	1,000,000	49,999,999	1, 4, 8
IN	1	100,000	999,999	8
KY	3	100,000	9,999,999	8, 12
LA	4	100,000	9,999,999	8, 12
MO	1	100,000	999,999	8
MS	2	100,000	999,999	8
NC	1	10,000	99,999	7, 8
ND	1	100,000	999,999	12
NE	1	10,000	99,999	9, 12
NV	1	10,000	99,999	12
OH	3	100	99,999	12
OK	1	100,000	999,999	12
OR	1	1,000,000	9,999,999	8
PA	3	100,000	9,999,999	8
SC	2	1,000	49,999,999	8, 12
TN	2	100,000	9,999,999	1, 4, 8, 9
TX	7	1,000	9,999,999	7, 8, 12
UT	1	10,000	99,999	9, 12
VA	2	100,000	49,999,999	8

Table 5-1. Facilities that Produce, Process, or Use Creosote

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
WI	2	1,000,000	9,999,999	8
WV	2	100,000	999,999	8

[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: TRI22 2024 (Data are from 2022)

The EPA is conducting a review of the Registration Eligibility Decision (RED) for creosote and issued a Preliminary Work Plan in March of 2015 (EPA 2015) and Registration Review Draft Risk Assessment in 2019 (EPA 2019). An Interim Registration Review Decision was released by the EPA in December 2020 (EPA 2020a). The last completed RED for creosote occurred in 2008. The EPA conducts reviews of registered pesticides every 15 years to determine under what conditions and uses they may continue to be used. In the Interim RED, the EPA reported that as of February 2020, there were 15 actively registered products containing coal tar creosote. According to the National Pesticide Information Retrieval System (NPIRS), there are five corporations that produce 16 different restricted use coal tar creosote products. Table 5-2 summarizes these data and lists currently registered products.

Table 5-2. Manufacturers of EPA Restricted Use Coal Tar Creosote Products

Company	Product Name	EPA Registration Number	Percent active ingredient
Arbor Preservative Systems, LLC 1421 Channel Avenue Memphis, Tennessee	Creosote	97080-6	98.5
	Creosote solution	97080-7	97.0
	Creosote for pressure application	97080-8	55.0
Coopers Creek Chemical Corporation 884 River Road West Conshohocken, Pennsylvania	The C-4 brand black creosote coal tar solution	363-14	95.0
	The C-4 brand coopersote creosote oil	363-15	98.5
	P-2 creosote-petroleum solution	363-48	75.0

Table 5-2. Manufacturers of EPA Restricted Use Coal Tar Creosote Products

Company	Product Name	EPA Registration Number	Percent active ingredient
Koppers Inc. 436 Seventh Avenue, K-1900 Pittsburgh, Pennsylvania	Coal tar creosote	61468-1	98.0
	60/40 creosote-coal tar solution	61468-3	95.0
	Creosote manufacturing use	61468-6	98.5
	Creosote petroleum solution	61468-9	75.0
Lone Star Specialty Products PO Box 247 Lone Star, Texas	Creosote solution	82024-1	97.5
	Creosote oil	82024-2	98.5
	P3 creosote petroleum solution	82024-3	75.0
Rain CII Carbon LLC 1330 Greengate Drove Suite 300 Covington, Louisiana	Coal tar creosote	61470-1	98.0
	Coal tar creosote P2	61470-3	98.0
	P3 creosote-petroleum solution	61470-4	75.0

Source: NPIRS 2022

In 2004, U.S. consumption of coal tar creosote was estimated at 785 million pounds (EPA 2008, 2015). A study conducted by the Treated Wood Council (TWC) estimated that approximately 82.9 million gallons (760 million pounds) of coal tar creosote were used in the United States in 2007 to treat 101 million cubic feet of wood (Bolin and Smith 2013). Data from the EPA Chemical Data Reporting (CDR) database showed a production volume of 6,190,222 pounds of coal tar creosote in 2019; however, the data only contained information from Lone Star Specialty Products (EPA 2022b). There were no reported production volumes for wood creosote (CAS Registry Number 8021-39-4).

5.2.2 Import/Export

The EPA CDR showed import volumes of coal tar creosote as 4,345,214 pounds in 2019 and no export volumes (EPA 2022b). The U.S. International Trade Commission (USITC) reported that 89,280,999 liters of creosote oils (HTS 27079100) were imported into the United States for consumption in 2021 with no domestic export data available (USITC 2022).

5.2.3 Use

Coal tar creosote has been used as a wood preservative pesticide in the United States for over 100 years. It is a fungicide, insecticide, and sporicide used as a wood preservative for above- and below-ground wood protection treatments as well as for treating wood in marine environments and each of the currently registered products in Table 5-2 are restricted use pesticides meaning that they are not available for purchase by the general public in the United States and may only be used by certified pesticide applicators (EPA 2008, 2015). Coal tar creosote products are registered for use in the pressure treatment of terrestrial and aquatic non-food wood and wood structures. According to EPA, there are two major types of coal tar creosote for use as a pesticide (EPA 2008, 2015). The P1/P13 fraction is used in the pressure treatment of utility poles and pilings. The P2 fraction is used in the pressure treatment of railroad ties/crossties and is more viscous than the P1/P13 blend. Potential end uses include utility poles/crossarms, railroad ties, switch ties, bridge timbers, fence and guardrail posts, foundation timbers, marine and foundation round piles, sawn lumber and timber products, and exterior structural composite glue laminated wood and plywood products. There are no registered residential uses of coal tar creosote or creosote-treated wood (EPA 2020a). Coal tar is used in the production of coal-tar products, such as coal tar creosote and coal-tar pitch, and refined chemicals. Low concentrations of coal tar have long been used to treat various skin conditions, such as eczema, psoriasis, and dandruff (NCI 2018; Veenhuis et al. 2002). The 2020 CDR contains many different entries of coal tar distillates all with unique CAS Registry Numbers that are products derived from coal tar under different distilling conditions. The major use of coal tar pitch is as the binder for aluminum smelting electrodes. Pitch is also used in roofing, surface coatings, and pitch coke production. Pipe-coating enamels made from pitch are used to protect buried oil, gas, and water pipes from corrosion (IARC 1985). Carbon black is also produced from the combustion of coal tar.

Beechwood creosote and its compounds, calcium creosotate, creosote carbonate, and creosote valerate, were used in the past as antiseptics and expectorants (Budavari 1989). Treatments for leprosy (Samson and Limkako 1923), pneumonia (McKinlay 1933), and tuberculosis (Fellows 1939a) also involved ingestion of beechwood creosote. Beechwood creosote is rarely used in the United States for medicinal purposes today.

5.2.4 Disposal

According to the TRI, 44,835 pounds of coal tar creosote were transferred off-site from facilities that use or process coal tar creosote, presumably for treatment and disposal (TRI22 2024). Treatment of creosote sludge generated from coal tar creosote production includes fixing, solidifying, and covering with clay. In the past, settling lagoons were used in treatment. However, they are no longer being used, and those which were used are now being remediated. "Disposal in place" requires groundwater monitoring for a

30-year period (Ball et al. 1985). Four Resource Conservation and Recovery Act (RCRA) hazardous wastes are listed due, in part, to their creosote content (40 CFR 261.31 and 261.32 [EPA 1981a, 1981b]):

- Waste waters, process residuals, preservative drippage, and spent formulations from wood preserving processes generated at plants that use creosote formulations
- Bottom sediment sludge from the treatment of waste waters from wood preserving processes
- Wastewater treatment sludges generated in the production of creosote
- Off-specification creosote (does not meet desired chemical composition).

Due to RCRA Land Disposal Restrictions, creosote can no longer be disposed in hazardous waste landfills unless it meets EPA specified treatment standards (EPA 1990). No technology- or concentration-based standards for the three RCRA hazardous wastes containing creosote specify creosote as a constituent for monitoring treatment performance (40 CFR 268.43 [EPA 1988a]). Industrially used creosote-treated wood can be burned in an industrial incinerator or boiler (EPA 1986a). Treated wood used in the home or farm should be buried or disposed with household garbage; it should not be incinerated (AWPA 1988). The potential for many types of hazardous pollutants to be included with creosote wastes seriously diminishes the potential for recycling or re-use.

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 2022c). 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's North American Industry Classification System (NAICS) codes is covered under EPCRA Section 313 or is a federal facility; and if their facility manufactures (defined to include importing) or processes any TRI chemical in excess of 25,000 pounds, or otherwise uses any TRI chemical in excess of 10,000 pounds, in a calendar year (EPA 2022c).

There are no known natural sources of creosote mixture (IARC 1973). However, several of the PAH constituents of creosote mixtures are known to have natural sources; the reader is referred to the ATSDR toxicological profile for PAHs (ATSDR 1995) and cresols (ATSDR 2008a) for additional information on natural sources, releases, and levels of PAHs and cresols associated with creosote production, use, and disposal.

PAH levels in environmental media are typically used as a metric for coal tar creosote releases from nearby point sources such as wood treatment facilities. However, levels in these media are confounded by the many sources of PAHs in the environment including vehicle emissions, coke-oven emissions, and coal, oil, and wood combustion that result in atmospheric deposition of PAHs to water, soil, sediment, and vegetation. PAH levels near a known source (e.g., wood treatment facility using coal tar creosote) are most reflective of releases from that source.

Spills from wood treatment facilities or wastewater effluents are a major source of creosote released to the environment (IPCS 2004). Emissions may also occur during the transfer of creosote from an incoming tanker or rail car to plant storage facilities (EPA 1998). Transfer of the product, whether from rail car or tanker, is typically performed using a closed piping system. The greatest chance for fugitive emissions is at the origin, where creosote is leaving the tanker or rail car, and at the end of the transfer, where creosote is entering the storage vessel. Coal tar creosote components may also be slowly released from the surface of treated wood products by oil exudation, leaching by rainwater, or volatilization. Losses of creosote from impregnated wood are dependent on temperature, salinity, water flow, density of the wood, and length of time since treatment of the wood (CSCC 2010). Kang et al. (2005) studied the leaching behavior of creosote treated wood in flowing fresh water by monitoring PAH levels over time at varying flow rates. Seven of 16 monitored compounds were detected in the water, and all detected PAHs increased immediately after immersion, then decreased sharply, and reached a steady state after 1 week. Higher molecular weight PAHs, including anthracene, chrysene, benzo[b]fluoranthene, benzo[f]fluoranthene, benzo[a]pyrene, indenopyrene, benzo[e]perylene, and dibenzo[a,h]anthracene, were not detected at any point in the test. The results suggest that PAH concentrations from creosote-treated wood appear to decline rapidly to ng/mL levels after initial exposure.

Treatment of wastewaters from wood-preserving processes that use creosote and/or pentachlorophenol produces bottom sediment sludge. EPA defines these as K001 sludges (EPA 1980); in the early 1990s, approximately 1,000 metric tons per year of K001 sludges were produced from active wood-preserving facilities (Davis et al. 1993). At that time, 55 wood-preserving facilities had been identified as NPL sites primarily because of contamination with K001 sludge (Davis et al. 1993).

Creosote-containing materials are also encountered at abandoned dump sites or abandoned facilities where creosote was produced or used in significant amounts. In addition to wood-preserving facilities, coal tar creosote was a byproduct of the production of so-called town gas, an illuminating gas made from coal (Arvin and Flyvbjerg 1992; EPA 1988b; Flyvbjerg et al. 1993). Around the turn of the century, virtually every large community in the United States had such a manufactured gas facility (EPA 1988b). From 1816 to 1947, more than 11 billion gallons of coal tar were generated at manufactured gas plants in the United States (Lee et al. 1992). The total number of town-gas sites may have approached 11,000. Several hundred of the larger sites have been evaluated for the NPL. Coke-producing facilities also generate coal tar wastes, including cresol emissions to the atmosphere (Grosjean 1991).

At older production facilities or places where wastes have been disposed off-site, the creosote materials are often mixed with other chemicals. For instance, pentachlorophenol (PCP) is commonly encountered at NPL sites involved with wood-preserving operations along with such metals as copper, chromium, and arsenic (Davis et al. 1993; Kuehl et al. 1990; Mueller et al. 1989, 1991). At many of these sites, PAHs from combustion sources other than coal tar may have been introduced. The wastes from old town-gas sites may contain benzene, toluene, ethylenebenzene, or xylenes, and sometimes cyanides (Arvin and Flyvbjerg 1992; EPA 1988b; Flyvbjerg et al. 1993).

No major sources of wood creosote releases to the environment have been reported.

5.3.1 Air

Estimated releases of 145,457 pounds (~66 metric tons) of creosote to the atmosphere from 55 facilities reporting to TRI domestic manufacturing and processing facilities in 2022, accounted for about 36% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). These releases are summarized in Table 5-3.

Table 5-3. Releases to the Environment from Facilities that Produce, Process, or Use Creosote[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
AL	5	9,210	46	0	0	0	9,256	0	9,256
AR	4	28,236	0	0	0	0	28,236	0	28,236
CA	1	3	0	0	160,506	0	160,509	0	160,509
CT	1	454	0	0	0	0	454	0	454
DE	1	15	0	0	2,529	0	99	2,445	2,544
IL	2	6,500	0	0	0	0	6,500	0	6,500

Table 5-3. Releases to the Environment from Facilities that Produce, Process, or Use Creosote[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	5,528	0	0	0	0	5,528	0	5,528
KY	3	5,486	0	0	13,249	0	10,246	8,489	18,735
LA	4	33,659	24	0	10,950	0	44,361	272	44,633
MS	2	2,885	43	0	0	0	2,928	0	2,928
MO	1	5,532	0	0	0	3,568	5,532	3,568	9,100
NE	1	4	0	0	0	0	4	0	4
NV	1	0	0	0	30,267	0	30,267	0	30,267
NC	1	0	0	0	0	5,880	0	5,880	5,880
ND	1	0	0	0	0	0	0	0	0
OH	3	1	0	0	0	0	1	0	1
OK	1	40	0	0	0	0	40	0	40
OR	1	604	0	0	0	0	604	0	604
PA	3	6,911	18	0	4,335	0	6,929	4,335	11,264
SC	2	2,328	0	0	66	0	2,328	66	2,393
TN	2	4,106	0	0	397	0	4,106	397	4,503
TX	7	13,322	22	0	21,683	0	17,767	17,261	35,028
UT	1	0	0	0	0	0	0	0	0
VA	3	10,373	13	0	0	0	10,386	0	10,386
WV	1	2,623	0	0	0	2,122	2,623	2,122	4,745
WI	2	7,637	13	0	8	0	7,658	0	7,658
Total	55	145,457	179	0	243,990	11,570	356,361	44,835	401,196

[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, wastewater 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: TRI22 2024 (Data are from 2022)

Coal tar creosote constituents such as naphthalene, acenaphthalene, acenaphthene, phenanthrene, and fluorene have been detected in emissions at a pressure treatment facility that treated logs for use as utility poles and marine pilings (EPA 1986b). Releases may occur at several points in the treatment process, such as when cylinder doors are opened after a treatment cycle, or when creosote is transferred from the heater to the cylinder at the beginning of the impregnation process. Atmospheric releases vary from plant to plant, depending on the process design, and are significantly smaller than releases to surface water in aqueous effluents (Henningsson 1983). It should be noted, however, that the more volatile PAHs may be less toxic (and especially less carcinogenic) than the less volatile PAHs.

Gallego et al. (2008) examined the emissions of volatile organic compounds (VOCs) and PAHs from wood recently treated with creosote. The primary components of the vapors released from the creosote-treated wood were identified as naphthalene, toluene, *m*-/*p*-xylene, ethylbenzene, *o*-xylene, isopropylbenzene, benzene, and 2-methylnaphthalene. VOC emission concentrations ranged from 35 mg/m^3 of air on the day of treatment to 5 mg/m^3 8 days later. PAHs emission concentrations were 28 μg/m^3 of air on the day of treatment and 4 μg/m^3 8 days later. Volatilization is likely to be greater during warmer months when ambient temperatures are higher. Gevao and Jones (1998) observed greater volatilization of acenaphthene, fluorene, phenanthrene, anthracene, and fluoranthene from creosote-treated wood at 30°C than at 4°C.

Volatilization from coal tar-based pavement sealants has been identified as a source of VOCs and PAHs into the atmosphere. Van Metre et al. (2012) measured PAH levels above parking lots sealed with coal tar-based sealants and compared them to levels of unsealed or asphalt lots. The geometric mean concentration of the sum of eight frequently detected PAHs 0.03 m above the lots using coal tar-based sealants was 1,320 ng/m^3. This was approximately 20 times greater than the total PAH levels in the unsealed lots (66.5 ng/m^3).

Coal tar is listed as a pollutant in the National Emissions Inventory (EPA 2017a). EPA's NEI database contains data regarding sources that emit criteria air pollutants and their precursors, and hazardous air pollutants (HAPs) for the 50 United States, Washington DC, Puerto Rico, and the U.S. Virgin Islands (prior to 1999, criteria pollutant emission estimates were maintained in the National Emission Trends [NET] database and HAP emission estimates were maintained in the National Toxics Inventory [NTI] database). The 2017 NEI report lists coal tar as a HAP, with air emissions ranging from 0.2 pounds (waste disposal) to 4,407 pounds. The industrial sectors and emissions for 2017 reporting are shown in Table 5-4.

Table 5-4. Reported Emissions from the 2017 NEI for Coal Tar

Sector	Pollutant type	Emissions (pounds)
Industrial processes; storage and transfer	HAP	4,407.2
Industrial processes; chemical manufacture	HAP	4,301
Solvent; industrial surface coating and solvent use	HAP	2,048.2
Industrial processes; not elsewhere categorized	HAP	1,384.6
Solvent; graphic arts	HAP	192
Industrial processes; petroleum refineries	HAP	102
Bulk gasoline terminals	HAP	73.6
Waste disposal	HAP	0.2
Industrial processes; non-ferrous metals	HAP	0

HAP = hazardous air pollutant

Source: EPA 2017a

Other potential sources of atmospheric releases include incineration of scrap wood treated with the mixture and re-entrainment of dust and soils contaminated with components of the mixture in the vicinity of hazardous waste sites.

5.3.2 Water

Estimated releases of 179 pounds (<1 metric tons) of creosote to surface water from 55 domestic manufacturing and processing facilities in 2022, accounted for <1% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). This estimate includes releases to wastewater treatment and publicly owned treatment works (POTWs) (TRI22 2024). These releases are summarized in Table 5-3.

A source of coal tar creosote released into surface waters and ground water is wastewater effluents from wood-preserving facilities or accidental spills (IPCS 2004). In previous years, wastewater generated from wood treatment facilities was often discharged to unlined evaporation/settling lagoons where a sludge was formed. Water-soluble coal tar creosote components then percolated through the soil to reach the groundwater table. Waste waters may include process water generated from steam conditioning of the wood; preservative formulation recovery and regeneration water; water used to wash excess preservative from the surface of the wood; condensate from drying kilns used to dry preserved or surface-protected wood; water that accumulates in door and retort sumps; and rain falling on or in the immediate vicinity of the treating cylinder and work tank area. Groundwater contamination from coal tar creosote waste waters and sludge stored in unlined surface water impoundments occurred at a wood treatment facility in Pensacola, Florida (Baedecker et al. 1988; Elder and Dresler 1988; Goerlitz et al. 1985). Similar contamination problems have occurred in Conroe, Texas (Borden 1986), and St. Louis Park, Minnesota (Hickok et al. 1982). An additional source of coal tar creosote released to waters is due to leaching from coal tar creosote-treated wood pilings (CSCC 2010). Leaching rates of contaminants from coal tar creosote-treated wood are variable and greatest during the first few years after placement, but also continues for many years. Leaching from coal tar creosote treated wood pilings is a function of salinity, temperature, flow, density of the wood, length of time since treatment of the wood, whether leaching occurs from the end grain or the face, and the surface area-to-volume ratio. In an investigation of the release of coal tar creosote from treated wood into fresh water and sea water, naphthalene, phenanthrene, acenaphthene, dibenzofuran, fluorene, and 2-methylnaphthalene were found to be the major components that migrated into water (Ingram et al. 1982). The rate of migration was found to increase significantly with increasing temperature within the range of 20–40°C; slower migration occurred from aged than from freshly treated pilings. In a microcosm study of the leaching of PAHs from coal tar creosote-impregnated pilings into aquatic environments, the aqueous concentration of PAHs increased with the number of pilings used (Bestari et al. 1998). The study authors calculated a rate loss of coal tar creosote from the wood pilings into the water of approximately 50 $\mu g/cm^2/day$ (273 mg/piling/day). Coal tar creosote was observed to be removed from the water rapidly after 7 days and was close to background concentrations (0.8–6.7 μg/L) by 84 days; losses were attributed to photolysis and microbial degradation, while sorption to sediment was not significant.

Given the very viscous nature of coal tar creosote or coal tar creosote-containing wastes, significant migration into groundwater supplies is seldom encountered unless the soils are extremely porous. For instance, a very sandy substrate at the American Creosote Works NPL site at Pensacola, Florida, allowed a significant plume of wood-preserving wastes to enter the ground water (Goerlitz et al. 1985). In most instances, the main concern over coal tar creosote materials entering well water is that minute quantities (ng/L) of coal tar components produce extremely objectionable tastes and odors (Arvin and Flyvbjerg 1992).

In addition to discharges or migration into ground water from disposal sites, coal tar creosote has often been introduced to receiving waters as the result of spills from wood treatment facilities or during the transportation of coal tar materials on barges or during loading and unloading accidents around docks or navigation facilities. Well-documented examples include a spill near Slidell, Louisiana, on the Bayou Bonfouca (DeLeon et al. 1988). During the years 1986–1991, 1,400 incidents of chemical and petroleum spills into the Newark Bay were documented; among these were spills of 53,000 gallons of liquid asphalt and 75 gallons of coal tar creosote (Gunster et al. 1993).

Runoff from coal tar-based driveway and parking lot sealants has been identified as a source of PAHs in nearby waters (Mahler et al. 2012). An estimated 85 million gallons (321 million liters) of coal-tar-based sealcoat are used annually in the United States.

5.3.3 Soil

Estimated releases of 243,990 pounds (~111 metric tons) of coal tar creosote to soil from 55 domestic manufacturing and processing facilities in 2022, accounted for about 61% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). These releases are summarized in Table 5-3.

In addition to accidental spills, coal tar creosote may be released to soils at wood treatment facilities because of bleeding of the product from treated timber in stockyard and storage areas. Rainwater may also wash the soluble components directly from the surface of treated timber and into the soil (Henningsson 1983). Localized, but severe, contamination of soils is often encountered on the grounds of older (often abandoned) wood-preserving or town-gas facilities (Davis et al. 1993; EPA 1988b). Coal tar creosote-treated wood pilings also may release constituents to sediment in a marine environment.

Dust and runoff from coal tar sealed driveways and parking lots is also a source of PAH contamination to urban waterways and sediments in locations where these products are used, although quantifying the direct contributions from sealants versus atmospheric deposition from other sources is subject to uncertainty (O’Reilly et al. 2011). Analysis of parking lot dust samples from six central and eastern U.S. cities where coal tar sealants are frequently used were shown to be much greater than dust samples in three western cities, which predominantly used asphalt-based sealants (Van Metre et al. 2009). The study authors found that bottom sediments of lakes of central and eastern U.S. cities contained greater PAH levels than the lakes of the western cities sampled and concluded that coal tar-based sealants were a source for these higher levels (Van Metre et al. 2009). A second study analyzed PAH levels in sediments from 40 urban lakes in areas in eastern and central U.S. versus levels from western lakes (Van Metre and

Mahler 2010). The study authors employed a chemical mass balance model to estimate the source apportionment of PAHs into the lake sediments and concluded that approximately 57% of the PAHs in bottom sediments in eastern and central U.S. lakes studied could be attributed to coal tar treated pavement, 11–20% arose from vehicle emissions, 18–26% from coal/oil combustion, and 5% from wood combustion.

5.4 ENVIRONMENTAL FATE

As with other chemical mixtures, the fate and transport processes affecting coal tar creosote can be extremely complex. The components of this mixture may partition to the air, water, soil, or biota depending on their physical and chemical properties. Compounds initially released to the atmosphere may undergo atmospheric deposition and reach surface water directly or through runoff carrying soil-bound compounds (Stangroom et al. 1998). For coal tar creosote materials encountered in old production facilities or waste disposal sites, materials contained in the top several feet of soil will have become "weathered," with virtually all the phenolic and heterocyclic fractions having volatilized, oxidized, or biodegraded (von Burg and Stout 1992). The lighter fractions of the PAH materials will also have degraded. The remaining weathered coal tar creosote will show limited ability to move off-site. Johnston et al. (1993) studied the PAH composition of coal-tar-containing samples collected at several coal gasworks sites in Australia. Most of these sites were abandoned nearly a century ago. The samples were taken from areas where the coal tar components would have undergone environmental modification to varying degrees since deposition. They concluded that aqueous partitioning and volatilization are probably the main processes that control environmental modification of coal tar at gasworks sites. As with releases to water, the migration of newly sealed and weathered driveway and parking lots that use coal tar-based driveway sealant may be a source to nearby soils (Mahler et al. 2012).

Newly produced coal tar creosote, or materials from a spill or a more recent disposal site, may pose more serious toxicity concerns. A complicating factor in interpreting the available literature is that coal tar creosote alone may not by the only source of toxicity. Especially at NPL or other waste disposal sites, such chemicals as pentachlorophenol (PCP) or heavy metals may be involved. Without an extensive battery of chemical analyses, perhaps combined with bioassay tests, making even semi-quantitative judgements on toxicity issues can be problematic. Much of the remedial work conducted under the Superfund program has simply aimed to reduce the volume of wastes at NPL sites with coal tar creosote contamination. A large percentage reduction by total weight does not always translate into a corresponding reduction in toxicity (Brooks et al. 1998; Hyötyläinen and Oikari 1999; Mueller et al. 1991).

5.4.1 Transport and Partitioning

Air. The environmental fate and transport of wood creosote, coal tar creosote, coal tar, and coal tar pitch are reflective of the individual components of these complex mixtures. Some identifiable components of these mixtures were presented in tables in Chapter 4. Phenols, which are representative of the components of wood creosote, generally have moderate to high vapor pressures and would be expected to exist primarily in the vapor phase in the ambient atmosphere. For example, phenol has a vapor pressure of approximately 0.35 mmHg at 25°C (EPA 2012). These compounds typically have relatively short atmospheric half-lives so would not be subject to long range transport in air. Coal tar creosote, coal tar, and coal tar pitch are more complex chemical mixtures; however, the lower molecular weight substances are also semi-volatile and tend to exist in the vapor phase in the ambient atmosphere. According to the International Programme on Chemical Safety chemical assessment of coal tar creosote, there are six major classes of compounds present in most mixtures: aromatic hydrocarbons, including PAHs and alkylated PAHs (which can constitute up to 90% of coal tar creosote); tar acids/phenolics; tar bases/nitrogen-containing heterocycles; aromatic amines; sulfur-containing heterocycles; and oxygen-containing heterocycles, including dibenzofurans (IPCS 2004). In general, phenolic compounds, low molecular weight PAHs, and some heterocycles tend to exist predominantly in the gaseous phase; however, the higher molecular weight PAHs will likely be present predominantly in the particulate phase. Substances in the particulate phase generally have longer atmospheric half-lives than vapor-phase substances and are removed from the atmosphere by wet and dry deposition.

Water. Coal tar creosote constituents released to surface waters will differentially partition to the water column or to sediments depending on their water solubility and sorptive properties. For example, PAHs, the major constituents of coal tar creosote, generally tend to sorb strongly to soil and sediment particulates, and often have low aqueous solubilities and mobility (Hickok et al. 1982; IPCS 2004). Many components in the PAH fraction, particularly the higher molecular weight PAHs, will remain in a virtually stationary tar-like mass at the place where they were deposited (dense nonaqueous phase layer [DNAPL]). Nitrogenous bases present in coal tar creosote wastewater (e.g., aniline, toluidines, and xylidines) are relatively soluble, mobile, and persistent in groundwater (Pereira et al. 1983). Volatilization from water surfaces is likely only an important environmental fate process for phenols and low molecular weight PAHs. Behavior at a given site is also dependent on site-specific characteristics.

For example, PAHs, phenol, and heterocyclic components of coal tar creosote wood treatment process wastes were found to migrate *en masse* in groundwater through a contaminated sand and gravel aquifer in Pensacola, Florida; sorption of these different classes of organic constituents in the low organic carbon (<0.1%) aquifer materials was not important (Pereira and Rostad 1986).

Similar to other environmental fate and transport properties, the potential to bioconcentrate and bioaccumulate in aquatic organisms is highly dependent upon the properties of the individual constituents of the complex mixtures. Jonsson et al. (2004) studied the bioconcentration potential of eight different PAHs representative of coal tar creosote mixtures in fish (sheepshead minnows) at two different exposure levels using a flow through aquarium. Bioconcentration factor (BCF) values in the fish ranged from 145 (pyrene) to 23,859 (2-isopropylnaphthalene) in the low-exposure (7.57 μg/L) group and from 97 to 46,536 in the high-exposure (72.31 μg/L) group.

Sediment and Soil. Sediment and soil tend to act as an environmental sink for most constituents of coal tar creosote, coal tar, and coal tar pitch, particularly the high molecular weight PAH components. The rate of vertical or horizontal migration of these components in soil is dependent upon the physical-chemical properties of the individual components of the mixture as well as the soil properties and environmental conditions (IPCS 2004). Laboratory model and field experiments (simulating coal tar creosote spills) showed a high retardation of transport of high molecular weight compounds coupled with a fast downward migration of lower molecular weight compounds. In an investigation of the partitioning of PAHs from coal tar wastes at manufactured gas plant sites into groundwater, partitioning of the various fractions of the complex mixture was observed to be inversely related to solubility, with the more soluble compounds partitioning to water more readily (Lee et al. 1992).

In a study of the extent of coal tar creosote contamination at four wood-preservative plants with process water surface impoundments, unspecified coal tar creosote components were found to have moved 20–60 feet vertically from the impoundments to the water table and up to 500 feet horizontally from the sources (Ball 1987). In a 50-day microcosm study of the aquifer materials of the Libby, Montana, Superfund site, 59% of radiolabeled phenanthrene was bound to the soil, while only 2.2% was volatilized (Mohammed et al. 1998).

In an investigation of the volatilization of PAHs from coal tar creosote-treated wood, desorption of acenaphthene, fluorene, phenanthrene, anthracene, and fluoranthene was directly related to concentration and was greater at 30°C than at 4°C (Gevao and Jones 1998). The study authors reported desorption half-lives of 0.7–31 years at 4°C and 0.3–1 year at 30°C for fluoranthene and acenaphthene, respectively. It is also possible to have volatilization from surface soil to the atmosphere. Coal tar constituents have Henry's law constants ranging from 0.11 to 8.65×10^{-8} atm m^3/mole and vapor pressures of 95 to 1.2×10^{-8} mmHg (Swann et al. 1983), indicating that some newly leached compounds may volatilize from both moist and dry soil surfaces.

In a terrestrial microcosm study that examined the transport of coal tar creosote containing substances impregnated in wood posts, 2.7% of radiolabeled phenanthrene and 4.3% of radiolabeled acenaphthene were found in soil samples taken in a 10-cm zone around coal tar creosote-treated posts, whereas concentrations of the compounds that remained in the posts were 95 and 93.5% of the amounts applied, respectively, after 2.5 months (Gile et al. 1982).

In an investigation of coal-tar contaminated surface sediments, PAHs were observed to have moved 400 m in groundwater from buried subsurface coal tar; persistence of the PAHs, naphthalene in particular, was partially attributed to anoxic conditions (Madsen et al. 1993, 1996). Additionally, sediment-bound coal tar creosote components may be released over time. In a laboratory study of coal tar creosote-contaminated sediment and natural lake water, Hyötyläinen and Oikari (1999) found that coal tar creosote-derived 4–6-ring PAHs released from the sediment during incubation were toxic to water fleas (*Daphnia magna*) and to the photoluminescent bacteria *Vibrio fischeri*.

Other Media. Atmospheric deposition of VOCs and PAHs contained in coal tar creosote is a possible source of contamination of leafy parts of plants and vegetable; however, uptake from roots is also possible. Moret et al. (2007) studied levels of light molecular weight PAHs (fluorene, phenanthrene, anthracene, fluoranthene, pyrene) and heavy molecular weight PAHs (benz[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, dibenz[a,h]anthracene, benzo[g,h,i]perylene, indeno[1,2,3-cd]pyrene) in olives grown in locations that stored old railroad ties that were treated with coal tar creosote. Levels of the light-weight PAHs were approximately 5,679 and 6,359.9 μg/kg in olive oil extracted from trees grown 1 and 2 m away from the railroad ties, respectively but oil extracted from trees 50 m away had total PAH levels of only 41.1 μg/kg. The study authors also indicated that very little of the heavy PAH components were found in olive oil extracts. The total concentration of heavy PAHs in olive oil from trees 1 m away was 6.2 μg/kg and decreased to 2.8 μg/kg at a distance of 50 m.

Animals such as voles, crickets, snails, pill bugs, and worms have exhibited the capacity to assimilate radiolabeled coal tar creosote components in terrestrial microcosm studies. Coal tar creosote components were found to accumulate to the greatest extent in the vole, with BCFs of 12–31. The ^{14}C mass balance content of the animals was 1.2% of applied acenaphthene and 0.8% of applied phenanthrene versus 4.3 and 2.7%, respectively, in soils (Gile et al. 1982). In addition, mussels taken from coal tar creosote-treated pilings have been found to contain significantly more benzo[a]pyrene, a coal tar creosote constituent, than those growing elsewhere (Dunn and Stich 1976). Accumulation of coal tar creosote-derived PAHs has occurred in benthic organisms in Pensacola Bay (Elder and Dresler 1988; Rostad and Pereira 1987). Fluoranthene, pyrene, benzo[a]pyrene, anthracene, chrysene, and phenanthrene were detected in higher concentrations in tissues of snails (*Thais haemastoma*) and oysters (*Crassostrea virginica*) taken from offshore sites near an onshore wood-treatment plant compared with those from control sites. The total PAH levels of mussels grown in cages near creosote treated wood pilings in Sooke Basin, British Columbia increased within 14 days of exposure from background levels of about ~16 to 68 ng/g, but then returned to baseline levels after 184 days (Brooks 2011a). No adverse effects on the survival of mussels suspended from the pilings were observed during the 2-year study period.

5.4.2 Transformation and Degradation

Air. Volatile constituents of these complex mixtures may undergo oxidation by vapor phase reaction with photochemically produced hydroxyl radicals, with calculated half-lives of 2 hours to 10 days based on experimental and estimated rate constants for representative coal tar creosote containing substances in the range of $1.12–103 \times 10^{12}$ cm/molecules-second at 25°C and using an average atmospheric hydroxyl radical concentration of 5×10^5 molecules/cm^3 (Atkinson 1989; Meylan and Howard 1993). Rates for constituents in the atmosphere with low vapor pressures may be slowed because they will exist in the particulate phase and, therefore, undergo atmospheric oxidation and direct photolysis at slower rates as compared to substances that exist primarily in the vapor phase (Eisenreich et al. 1981). Additionally, some components may undergo nighttime reactions with nitrate radicals (Atkinson et al. 1987). Based on an experimental rate constant of 3.8×10^{-12} cm/molecules-second for phenol, and an atmospheric nitrate radical concentration of 2×10^8 molecules/cm^3, a half-life of 15 minutes can be calculated for the compound (Atkinson 1989).

Water. Many of the constituents of these complex mixtures present in surface waters may be degraded by direct and indirect photolysis. Estimated aqueous photolysis half-lives of 8.4, 71, and 21 hours have been reported for phenanthrene, naphthalene and fluoranthene, respectively (Zepp and Schlotzhauer

1979). Other constituents which may undergo aqueous photolysis are acenaphthalene, anthracene, benzene, quinoline, phenol, cresol. In a microcosm study, PAHs leached from coal tar creosote-impregnated wood pilings were degraded in aquatic environments by photolysis and microbial degradation, while sorption to sediment was not significant (Bestari et al. 1998).

Coal tar creosote components are degraded in aquatic environments mainly by microfauna metabolism (Borthwick and Patrick 1982; Ingram et al. 1982). Microorganisms may act on the coal tar creosote-treated wood itself or on coal tar creosote components that have leached from the treated wood. Quinoline, the major tar base in coal tar creosote, may be degraded in surface water and ground water by bacteria of the genus *Pseudomonas* (Bennett et al. 1985). Biotransformation of the phenolic components of coal tar creosote apparently also occurs under anaerobic conditions in contaminated ground water (Ehrlich et al. 1983; Goerlitz et al. 1985). Adaptation of soil microorganisms to PAH contaminants in ground water originating from coal tar creosote treatment plant wastes has also been reported (Wilson et al. 1985).

Work on NPL sites has helped identify numerous bacteria and fungi that can biodegrade coal tar creosote materials. In addition to *Pseudomonas*, bacteria in the genus *Alcaligenes* can degrade phenolic compounds under aerobic conditions (Mueller et al. 1989). So long as the ground water is not completely anoxic, numerous soil microorganisms can degrade coal tar creosote materials. Work at NPL sites suggests that up to 90% of the coal tar creosote degradation is associated with biologically mediated processes. Although this can lead to an appreciable reduction in the quantity of the coal tar creosote materials, it is the phenolic and lower molecular weight PAHs that are degraded while the higher molecular weight PAHs that have been shown to resist biological attack may persist. In a study of biodegradation of coal tar creosote-contaminated ground water from the American Creosote Superfund Site, Mueller et al. (1991) observed a toxic and teratogenic response of inland silverside (*Merida beryllina)* embryos to the biotreated water at both 10 and 100% concentrations. They attributed the response to the cumulative effects of carcinogenic higher molecular weight PAHs that remained after 14 days of incubation. The higher levels of biodegradation observed for the lower molecular weight PAHs was attributed to their greater aqueous solubility and consequent greater bioavailability.

Work on town-gas sites in Europe has demonstrated that where nitrate levels are high, or where nitrate is supplied to ground water, various facultative bacteria can degrade coal tar components using the nitrate or nitrite as an electron acceptor (Flyvbjerg et al. 1993). In general, however, biodegradation under anoxic conditions appears to proceed very slowly for most constituents of coal tar creosote. Even when supplied with ample quantities of such electron acceptors as nitrates, half-lives >20 days were observed in laboratory microcosms for the anoxic biodegradation of dimethylphenol components in coal tar creosote, and cresol components showed little indication of significant disappearance unless the experiments were continued in excess of 90 days (Arvin and Flyvbjerg 1992).

Coal tar creosote components have been detected in surface water samples taken near a wood-treatment facility that ceased operation 30 years earlier (Black 1982). The coal tar creosote, which appeared to have permeated the sandy surface soils down to an impervious clay layer, was entering the river via seepages and springs. Weathering processes produced only minor constitutive changes in the coal tar creosote with relative losses of the lower molecular weight components. These changes probably reflected the greater volatility and solubilities of the 2–3 carbon ring PAHs.

Sediment and Soil. Smułek et al. (2020) studied the biodegradation of PAHs in soils contaminated by creosote oil of a railway sleeper treatment plant in Koźmin Wielkopolski in central Poland. A total of 10 soil samples were collected from three different boreholes across the polluted area and used for laboratory studies to identify potentially useful creosote degrading bacterial strains that could be isolated from these soil samples. The authors identified *Pseudomonas mendocina* and *Brevundimonas olei* as the most effective strains that were capable of degrading more than 60% of the total content of PAHs during a 28-day incubation period.

A remediation method combining biodegradation and electroosmosis showed enhanced degradation of PAHs from coal tar creosote-polluted soils (Niqui-Arroyo and Ortega-Calvo 2007). A predominantly clay and a loamy soil were studied that contained a mixture of PAHs typically present in coal tar creosote and amended with a surfactant and soil bacterium capable of degrading PAHs. In the loamy soil, 50% degradation of benzo[a]pyrene was observed after only 7 days, which was significantly greater than the degradation observed using electrokinetical flushing and bioremediation alone.

In a study of PAH-contaminated soil from the Reilly Tar Superfund Site in St. Louis Park, Minnesota, total EPA priority pollutant PAH concentrations were decreased 48–74% following treatment with one of four bioremediation technologies (Brooks et al. 1998). The remediation methods included bioslurry, biopile, compost, and land treatment. None of the four techniques tested was successful at removing the 5- and 6-ring higher molecular weight PAHs; however, it was suggested that compost and land treatment processes were the most effective treatment techniques.

PAHs from soil contaminated with coal tar creosote can also be removed by biodegradation, using fungi. A field study at a wood treatment facility located in Mississippi investigated the effects of solid-phase bioremediation using the white rot fungus, *Phanerochaete sordida* (Davis et al. 1993). This fungus has been shown to also biodegrade PCP, which has often become mixed with the wastes found at coal tar creosote production or disposal sites. The study authors observed 85–95% degradation of 3-membered PAHs and 24–72% loss of 4-membered PAHs following a 56-day treatment period; however, PAHs containing 5 or more rings were persistent. Byss et al. (2008) observed the white rot fungus, *Pleurotus ostreatus*, to be a more efficient coal tar creosote-degrading organism than *Irpex lacteus* in a laboratory-scale study.

Other Media. Very little information was found in the available literature on the transformation or degradation of coal tar creosote or wood creosote in animals or plants. FWS (1987) found that many aquatic organisms are able to rapidly metabolize and eliminate PAHs, the major constituents of the commercial mixture.

5.5 LEVELS IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to creosote depends, in part, on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of creosote 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 creosote 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. Current air sampling methods for semi-volatile substances employ two-stage sampling media, which includes a filter to collect particles and a sorbent material to collect vapors. Data collected historically using only a filter or a sorbent material most likely underestimated actual atmospheric levels and subsequent inhalation exposures. Due to the lipophilic nature of many of the components of this mixture, care should be given to storage and handling of samples to avoid adsorption to a storage vehicle, which could lead to inaccurate measurements.

Table 5-5 shows the typical limit of detections that are achieved by analytical analysis in environmental media for some important PAHs or VOCs expected to be present in creosote mixtures. An overview summary of the range of concentrations detected in environmental media is presented in Table 5-6. PAHs are expected to be found in all environmental media from a variety of sources; therefore, only levels that were found around known sources of coal tar, coal tar creosote, or coal tar pitch sources are discussed.

Often, total PAHs will be provided in monitoring data. This typically refers to the sum total of 16 substances designated High Priority Pollutants by the EPA; they include: naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, benzo[g,h,i]perylene, indeno[1,2,3-c,d]pyrene, and dibenz[a,h]anthracene (Hussar et al. 2012).

Table 5-5. Lowest Limit of Detection for PAHs and VOCs in Creosote Mixtures Based on Standards[a]

Media	Detection limit	Reference
Air	0.0054−4.4006 ng/m^3	EPA 2020b
Drinking water	0.033–0.66 ng/L	Aygun and Bagcevan 2019
Surface water and groundwater	0.06–5,7 μg/L (pore water) 0.013–0.64 μg/L (groundwater/waste)	EPA 2007b Method 8272 EPA 1986c Method 8310
Soil	0.020 μg/g	USDA 2004
Sediment	0.020 μg/g	USDA 2004
Whole blood	20 ng/mL ~2 ng/mL	Ramesh et al. 2015 Anderson et al. 2015

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

PAH = polycyclic aromatic hydrocarbon; VOC = volatile organic compound

Table 5-6. Summary of Environmental Levels of Creosote

Media	Low	High	For more information
Outdoor air (ng/m^3)	1.6	90	Section 5.5.1
Indoor air (workplace air) (μg/m^3)	<LOD	1,211	Section 5.6
Surface water	<LOD	>100,000 μg/L	Section 5.5.2
Groundwater	<LOD	>100,000 μg/L	Section 5.5.2
Sediment	0.074 mg/kg	15,000 mg/kg	Section 5.5.3
Food/fish	0.11 mg/kg	60.1 mg/kg	Section 5.5.4
Soil	0.39 mg/kg	657 mg/kg	Section 5.5.3

LOD = limit of detection

Detections of coal tar creosote in air, water, and soil at NPL sites are summarized in Table 5-7.

Table 5-7. Coal Tar Creosote 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)	13,400	6,570	160	4	4
Soil (ppb)	21,000	20,500	190	4	4
Air (ppbv)			No data		

[a]Concentrations found in ATSDR site documents from 1981 to 2022 for 1,868 NPL sites (ATSDR 2022). 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

Information regarding creosote constituents in ambient air are confounded by the fact that PAHs will arise from other sources such as automobile emissions or industrial processes (Brooks 2011b). A maximum concentration of 90 ng/m^3 has been reported for naphthalene at 2,000 m from a creosote using facility, with levels decreasing with distance from the plant (IPCS 2004). For example, the concentration of fluoranthene decreased from 64 ng/m^3 at 500 m to 1.6 ng/m^3 at 5,000 m away from the facility. Chen et al. (2002) performed air monitoring studies at a cleanup site of a gasification plant in Kingston, Ontario that contained large underground tanks of coal tar. The study authors measured concentrations of naphthalene, typically the most abundant PAH in coal tars, over a 3-week monitoring period. A total of 168 half-hour concentrations were determined at several different locations upwind and downwind of the site. The highest half hour average level of naphthalene measured was 250 μg/m^3, and in 45% of the samples collected, the level of naphthalene exceeded the Ontario Ministry of the Environment guideline of 36 μg/m^3. Air sampling was performed in Globeville, Colorado, a residential area located near the Koppers wood treating facility and two asphalt plants, after residents complained of an odor described as tar or asphalt became noticeable (Morgan et al. 2015). Several VOCs and PAHs were detected during these odor events but were not often detected in background samples or the suspected industrial sources. Naphthalene, dibenz[a,h]anthracene, benzo[g,h,i]perylene, and indeno[1,2,3-cd]pyrene were detected in the air at levels of 25.00, 2.70, 1.70, and 2.10, ppbv (131.06, 30.73, 19.21, and 23.73 μg/m^3), respectively, during one of these odor events. For example, naphthalene has a geometric mean odor detection threshold in air of 0.038 ppm (NLM 2023). ATSDR has been petitioned to evaluate exposures associated with many creosote facilities with odors being a common complaint (ATSDR 2004, 2007b, 2009). As a result, the agency developed a website to assist communities with their environmental odors (https://www.atsdr.cdc.gov/odors/).

Construction materials containing coal tar and coal tar creosote products may be a source of PAHs in indoor air. Kozicki and Niesłochowski (2020) published the results of indoor air measurements of 11 PAHs in 14 buildings including residential buildings, office buildings, and public buildings such as schools, hotels, and museums that used coal tar products to treat wood structures or other construction materials such as coal tar containing bituminous mixtures or coal tar containing adhesives. Indoor air levels tended to be greatest for naphthalene and methylnaphthalenes, with maximum levels of 42±6 and 34±5 μg/m^3, respectively, measured in public buildings. Six PAHs (naphthalene, methylnaphthalenes, dimethylnaphthalenes, biphenyl, acenaphthene, and dibenzofuran) were detected in residential, office, and public structures. Fluorene, phenanthrene, anthracene, fluoranthene, and pyrene were not detected in any of the five office buildings, or five public buildings studied; however, they were detected in residential buildings, with maximum levels of 3±1 μg/m^3 (fluoranthene) to 11±2 μg/m^3 (anthracene). Piñeiro et al. (2021) discussed PAH levels in the indoor air of a residential structure in Madrid, Spain constructed using waterproof coal tar membrane roofing materials. PAHs such as naphthalene, methylnaphthalenes, acenaphthene, acenaphthylene, phenanthrene, and fluorine were detected at levels that exceeded recommended indoor air guidelines. Total PAH levels of the 16 High Priority Pollutants were 1,167 μg/m^3 in the living room, and naphthalene levels extracted from the inter-joist pan form were as high as 6,152 μg/m^3.

ATSDR's guidance for evaluating the vapor intrusion pathway identifies coal tars and creosote as a potential source of volatile and semi-volatile contaminants, including naphthalene, benzene, toluene, ethylbenzene, and xylene (ATSDR 2016a). The contaminants may off gas from creosote and coal tar contamination in groundwater and soil gas and migrate up into the indoor air of buildings by a process called vapor intrusion. DNAPL may serve as an ongoing source of contamination into groundwater as contaminants dissolve into the aqueous phase.

Coal tar-based driveway and parking lot sealants have been associated with high levels of PAHs in indoor dust samples (Mahler et al. 2010). A study that examined PAH levels in the indoor dust from 23 apartments found that the median concentration of total PAHs in dust from coal tar sealed parking lots was 4,760 μg/g (n=11). The median indoor dust level from 12 residencies that had pavement surfaces not sealed with coal tar-based products was 9 μg/g.

Workplace air concentration data are discussed in Section 5.6. Data on ambient atmospheric concentrations of PAHs derived from other sources can be found in the ATSDR toxicological profile for PAHs (ATSDR 1995).

5.5.2 Water

Levels of PAHs in the ppb to ppm range have been found for some individual PAHs in surface water at creosote contaminated sites (IPCS 2004).

Following a fire at a wood treatment facility and subsequent creosote spill in Louisiana, PAHs were detected in surface water in Bayou Bonfouca. Levels of selected PAHs were: 400–39,700 µg/L anthracene; not detected–5,500 µg/L benzofluoranthenes; 300–6,600 µg/L benzo[a]pyrene; 1,200–110,000 µg/L fluoranthene; 600–12,300 µg/L fluorene; 700–14,100 µg/L naphthalene; 2,300–155,000 µg/L phenanthrene; and 2,100–85,000 µg/L pyrene (Catallo and Gambrell 1987). Sixteen PAHs were monitored in surface water from five railway ditches flowing to salmon streams in British Columbia, Canada (Wan 1991). At sites where PAHs were detected in ditch water, the average total PAH concentration was 606.9 µg/L, with a range of 1–3,515.9 µg/L.

The IPCS (2004) summarized groundwater levels of monocyclic aromatic and phenolic compounds detected at creosote-contaminated sites from eight different studies conducted in the United States, Canada, and Denmark. The highest level detected was for *m*-cresol (25,170 µg/L) at an abandoned wood treatment facility in the United States. This publication also summarized PAH levels at creosote contaminated sites in the United States, Canada, and Denmark from studies performed in the 1980s to the 1990s. The highest levels were observed in groundwater near the Escambia Wood Treating Company in Pensacola, Florida that manufactured treated utility poles, foundation pilings, and lumber with creosote and PCP from 1942 until 1982. Levels exceeded 100,000 µg/L for several PAHs including pyrene, phenanthrene, fluorene, and fluoranthene. This facility was abandoned in 1991 and remediation began shortly thereafter. EPA released the fourth 5-year review report and groundwater monitoring data conducted from 2013 to 2016 (EPA 2017b). Levels of most PAHs were significantly lower than previous measurements. For example, concentrations of phenanthrene in groundwater wells were below detection limits in wells sampled in 11 out of 12 sampling periods with a maximum concentration of 58 µg/L from sampling conducted in 2014. Fluorene was also not detected during most sampling periods and had a maximum concentration of 92 µg/L in sampling conducted in November 2014.

At the Koppers Company, Inc. NPL site in Texarkana, Texas, where a creosote wood treatment facility existed from 1903 to 1961, creosote-derived naphthalene, acenaphthene, fluorene, pyrene, and phenanthrene were measured in surface water and groundwater at levels up to 100,000 µg/L (ATSDR 1994). Remediation began in 1993 and the EPA 5-year review of this remediated site showed no surface water detections for 16 EPA criteria PAHs in sampling conducted from 2011 to 2014 (EPA 2016).

A public health assessment was conducted by ATSDR and the North Carolina Department of Health and Human Services Division of Public Health on the Holcomb Creosote Company that operated as a coal-tar creosote wood-treating facility from 1951 to 2009. EPA began remediation and removal actions to address contamination of environmental media at this site in 2011. Surface water samples collected after 2011 had onsite levels of total PAHs ranging from 7.14 to 31.8 µg/L (NC DHHS 2020).

Results from 2 years of groundwater sampling at an abandoned wood treatment facility in Conroe, Texas, where coal tar creosote had been used for about 20 years, showed that monitoring wells were contaminated with levels of up to 3,490 µg/L naphthalene, 1,263 µg/L methylnaphthalene, 425 µg/L dibenzofuran, and 302 µg/L fluorene. The contaminants had apparently migrated through the clay and sand soils on the site from three waste pits. A plume of groundwater contamination by organics at trace levels was found to extend up to 300 feet from the waste pit locations (Bedient et al. 1984).

5.5.3 Sediment and Soil

PAHs undergo a weathering process in soils and sediment (EPA 2006). This results in the lighter fractions (i.e., shorter chain molecules) being removed more readily than heavier PAHs. This occurs mainly by volatilization, but some proportion of the material moves through the soil vadose zone and into the groundwater. Heavier fractions tend to adsorb more readily to the soil organic matter and remain behind in the topsoil horizons. Weathering occurs in sediments as well, but much more slowly.

Soil samples were studied for PAH levels from the Des Plaines River wetlands in Will County, Illinois near a Commonwealth Edison railroad line (USDA 2004). PAHs were observed above detection limits in five of six baseline wetland soil samples. Total PAH concentrations ranged from 0.183 to 0.893 µg/g (183–893 µg/kg) dry soil with a mean and 95% confidence interval of 0.430±0.183 µg/g.

On- and offsite sampling was conducted at the Holcomb Creosote Company, North Carolina, which operated as a coal-tar creosote wood-treating facility from 1951 to 2009 (NC DHHS 2020). PAHs were detected in 18 of 23 onsite soil samples collected prior to 2011, with a range (total PAHs) of 0.39–290 mg/kg. Onsite sediment samples had detectable levels of PAHs in six out of seven samples, with summed total levels ranging from 7.05 to 657 mg/kg. Offsite sediment samples collected prior to 2011 had total PAH levels of 2.33–6.92 mg/kg (three out of seven positive samples).

Soil samples were studied at locations that stored old railway ties treated with coal tar creosote in Italy (Moret et al. 2007). High levels of PAHs were detected in soil samples very close to the railroad ties (0–1 m) with total concentrations of 2,157 μg/kg for light molecular weight PAHs (fluorene, phenanthrene, anthracene, fluoranthene, pyrene) and 3,121.8 μg/kg for heavy molecular weight PAHs (benz[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, dibenz[a,h]anthracene, benzo[g,h,i]perylene, indeno[1,2,3-cd]pyrene). The total PAH load was shown to decrease rapidly with the distance from the railway ties. For example, at a distance of 20 m, the level of the total light PAHs decreased to 15.2 μg/kg and the concentration of the total heavy PAHs was 25 μg/kg.

Baldwin et al. (2020) studied PAH contaminants in surficial streambed sediments at 71 locations across 26 Great Lakes Basin watersheds. Although there are numerous sources of PAHs to these watersheds, coal tar-sealed pavement dust was the most likely source of PAHs to most of the locations sampled. The total (summed) concentration of 16 EPA priority pollutant PAHs was 7.4–196,000 μg/kg (0.0074–196 mg/kg) and the median value was 2,600 μg/kg (2.6 mg/kg).

High levels of PAHs have been observed in the Elizabeth River near the Chesapeake Bay due to spills that occurred from the Atlantic Wood Industries facility (Di Giulio and Clark 2015). PAH levels in sediment samples remain high near wood treatment facilities years to decades after cessation of plant operations. A maximum total PAH concentration of 15,000 mg/kg was observed in sediments of the Elizabeth River adjacent to a wood treatment facility.

PAH contamination of soil has been found at the site of a wood-preservation facility that operated in Slidell, Louisiana, from 1892 to 1970, when a fire destroyed the plant facilities. It is believed that environmental releases of creosote occurred throughout the plant's operating history and as the result of the 1970 fire, when creosote was released from storage tanks and flowed over the ground and into adjacent water bodies. Waste creosote and debris have accumulated in eight areas at the site. The deposits are up to 2 feet thick and have contaminated underlying soils (based on visual inspection) to as much as 1 foot below the surface. PAH concentrations show a rapid decrease with increasing depth, ranging from 15,680 mg/kg (ppm) at the surface to 1 mg/kg (ppm) within 9 feet.

Several PAH constituents of creosote were detected in soil samples taken at an abandoned wood treatment facility in Conroe, Texas, at depths of up to 25 feet. Maximum concentrations of the compounds were detected in samples collected at the 0.7–1.8-foot depth. Maximum concentration levels were 3.7 mg/kg for naphthalene, 3.4 mg/kg for methylnaphthalene, 3.8 mg/kg for dibenzofuran, 4.2 mg/kg for fluorene, and 2.2 mg/kg for anthracene. An investigation of vertical variations in contaminant concentrations in the soil zone above the water table revealed that, in general, >90% of the organics were removed within the first 5 feet at the location studied. Organics can be degraded by microbes, adsorbed onto soil, or altered by interactions with soil humus (Bedient et al. 1984).

At the Koppers Company, Inc. NPL site in Texarkana, Texas, where a creosote wood treatment facility existed for 51 years prior to being converted to a residential area and an industrial site (sand and gravel company), creosote-derived pyrene, fluoranthene, phenanthrene, and anthracene (base/neutral compounds) were measured in surface and subsurface soils at levels ranging from nondetectable to 1,000 ppm (ATSDR 1994).

In sediment samples from a creek adjacent to the Koppers Company, Inc. NPL site, creosote-derived base/neutral compounds were detected at concentrations up to 100 ppm; one creosote-derived base/neutral compound was detected in downstream sediment at a maximum of 1 ppm (ATSDR 1994). Creosote-derived base/neutral compounds were also detected in the sediment of the drainage ditch at the site, at levels ranging from 1 to 100 ppm.

Coal tar creosote-derived phenanthrene, 1,2-benzanthracene, and benzo[a]pyrene have been detected in river sediments at concentrations of up to 231, 62, and 16 mg/kg (wet basis), respectively, directly downstream from the site of a former wood treatment facility. At 4,000 m from the source, these levels decreased to 0.35, 1.02, and 0.40 mg/kg (wet basis), respectively (Black 1982). Creosote-derived PAHs were also detected in the sediments of Pensacola Bay and a drainage stream in the vicinity of a former wood treatment facility near Pensacola, Florida. PAH concentrations ranged from 200 μg/g for naphthalene to 140 mg/kg for anthracene in stream sediments; concentrations in Pensacola Bay ranged from 75 μg/kg for benzanthracene to 190 μg/kg for fluoranthene (Elder and Dresler 1988).

PAH concentrations have been determined in sediment cores collected from the Arthur Kill, Hackensack River, and Passaic River in northern New Jersey. These rivers are in industrialized areas near former creosote wood-preserving facilities that operated through the 1960s and 1970s. Temporal distributions were determined in each core based on the activities of the radionuclides ^{210}Pb and ^{137}Cs. Sediments at depths corresponding to the years 1978 and 1964 contained total PAHs at concentrations of 1.71 mg/kg (ppm) for 1978, and not detected to 35.7 mg/kg for 1964 (Huntley et al. 1993). In a study of Eagle Harbor, an estuarine bay of the Puget Sound in which sediments were contaminated with creosote from a wood treatment facility, total PAHs were detected at concentrations as high as 6,461 mg/kg (Swartz et al. 1989).

5.5.4 Other Media

Fish and aquatic organisms exposed to coal tar creosote in waters have been shown to accumulate PAHs. Largemouth bass collected at Dobbins Pond (2012–2014) at the Holcomb Creosote Company, North Carolina had total PAH levels ranging from 0.11 to 60.1 mg/kg and white catfish had levels ranging from 0.057 to 0.07 mg/kg (NC DHHS 2020). Total PAH levels in largemouth bass from a reference pond not on the site had PAH levels of 0.121–0.221 mg/kg, and PAHs were not detected in any channel catfish samples off site. West et al. (2019) studied the potential accumulation of PAHs in Pacific herring (*Clupea pallasii*) embryos near creosote-treated pilings in the Puget Sound. Total PAH levels in embryo samples placed close to the creosote treated pilings were approximately 90 times greater than levels observed in reference embryo samples.

A study at a creosote spill near Lake Pontchartrain in Louisiana, provided some indications that biomagnification through food chains leading to humans can take place. This study documented the bioaccumulation of creosote-derived PAH fractions in the marsh clam *Rungia cuneata* (DeLeon et al. 1988). Clams introduced to an area near a major creosote spill showed tissue concentrations of benzopyrenes up to 600 ppb after 4 weeks of exposure, compared to a background level of 87 ppb. Marsh clams are a major food item for crustaceans, such as the blue crab, that are part of commercial fisheries in the Lake Pontchartrain area.

Olives grown near sites that stored old creosote treated railroad ties were shown to have PAHs levels which decreased with distance from the source and were different based upon the molecular weights of the PAHs studied (Moret et al. 2007). Levels of light weight PAHs (fluorene, phenanthrene, anthracene, fluoranthene, pyrene) were reported as 5,679 and 6359.9 µg/kg in olive oil extracted from trees grown 1 and 2 m away from the railroad ties, respectively, but oil extracted from trees 50 m away had total PAH levels of only 41.1 µg/kg, suggesting that the railroad ties were likely the source of PAHs in the olives. Levels of heavy PAHs (benz[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, dibenz[a,h]anthracene, benzo[g,h,i]perylene, indeno[1,2,3-cd]pyrene) components were much lower in olive oil extracts. The total concentration of heavy PAHs in olive oil from trees 1 m away was 6.2 μg/kg and decreased to 2.8 μg/kg at 50 m.

5.6 GENERAL POPULATION EXPOSURE

Since coal tar creosote is now a restricted use pesticide, the general population exposure to the PAHs associated with this use is limited to inhalation and dermal contact with potentially finished products. For persons residing near known sources exposures may be greater. Chemical contaminants from wood processing waste were reported in residents and residential homes adjacent to a wood treatment plant that used creosote and PCP to treat wood for over 70 years (Dahlgren et al. 2003, 2007). For a period of time, the plant also burned treated wood products. Analysis of blood samples from 10 residents showed elevated octachlorodibenzo-*p*-dioxin and heptachlorodibenzo-*p*-dioxin, consistent with PCP as the source. Soil sediment and dust samples had higher than background levels of carcinogenic PAHs. The estimated air levels for benzo[a]pyrene and tetrachlorodibenzodioxin were also elevated.

Potential sources of non-occupational human exposure to creosote include contact with creosote-treated wood products (e.g., railroad ties or poles), incineration of creosote-treated scrap lumber, and contact with contaminated environmental media at hazardous waste sites (e.g., ingestion of contaminated ground water). At the Koppers Company, Inc. NPL site in Texarkana, Texas, where a creosote wood treatment facility existed for 51 years prior to being converted to a residential area and an industrial site (sand and gravel company), a study by the Texas Department of Health found an increased incidence of skin rashes in residents who had dermal contact with soil at the site (ATSDR 1994). There is also potential for family members of workers from industries manufacturing or using coal tar or creosote products to be unintentionally exposed to the constituents of these mixtures from contaminated items such as worker clothing or footwear.

Risk of exposure to creosote constituents through contact with contaminated ground water will vary with the individual chemicals involved as well as with the mix of chemicals present at any one time and the environmental conditions. Physical and chemical properties of the compounds, including solubility and molecular weight, will affect distance the contaminant plume may travel from the source, as well as its susceptibility to biodegradation or sorption (King and Barker 1999). The environment in which contamination occurs is also of importance since natural attenuation of chemical compounds may be dependent on whether oxidizing or reducing conditions are present. In an investigation of natural attenuation of contaminant plumes from an emplaced coal tar creosote source, King et al. (1999) observed greater and more rapid decreases in plume mass for some compounds, such as phenol, *m*-xylene, and carbazole, while the dibenzofuran plume mass and extent remained relatively constant, and the plume mass and travel distance from the source for naphthalene and 1-methylnaphthalene increased throughout the 4-year study. Therefore, potential for exposure to creosote constituents present in groundwater will differ from location to location and over time.

Direct exposure of homeowners to wood treatment products containing creosote should be limited, since EPA has restricted the sale and use of such products to certified applicators. Industrial sources have noted that there have been no reports or instances of health effect allegations in the last 20 years (ending in 1996), except for rare reports of skin irritation resulting from public contact with creosote-treated wood.

Another potential source of nonoccupational exposure is the therapeutic use of coal tar shampoos for anti-dandruff therapy, coal tar ointments for treatment of eczematous dermatitis, and mineral coal tar for the treatment of psoriasis. Patients with atopic dermatitis and treated with topical coal tar preparations had increased urinary 1-hydroxypyrene excretion rates (Veenhuis et al. 2002). The urinary 1-hydroxypyrene excretion rate was dependent on the amount of coal tar applied to the skin and the total body area treated, and less on the severity of the atopic dermatitis. Adsorption of PAHs may occur through the skin, lungs, and gastrointestinal tract (Strickland et al. 1996). van Schooten et al. (1994) measured the urinary excretion of a specific PAH metabolite, 1-hydroxypyrene, to assess the internal dose of PAH after acute dermal application of coal tar shampoo. The shampoo selected for the experiment had a PAH concentration of 2,840 mg/kg, including pyrene (285 mg/kg) and benzopyrene (56 mg/kg). In other brands, the concentrations were at least 100 times lower. A single use of the coal tar shampoo resulted in increased 1-hydroxypyrene excretion in all participants. The mean increase of totally excreted 1-hydroxypyrene on day 1 was 10 times the pre-experiment background values. On day 2, the mean increase was 5 times. Interindividual variation was considerable, with a variation in the first day increase of between 3 and 20 times. The 1-hydroxypyrene values observed in coke oven workers are similar to the values obtained on day 1 after a single treatment with coal tar shampoo (0.4–8.3 μmol/mol creatinine) (van Schooten et al. 1994). However, exposure levels determined using the 1-hydroxypyrene biomarker may be affected by the time of measurement following exposure (Viau and Vyskocil 1995). Viau and Vyskocil observed maximum excretions of 1-hydroxypyrene in urine a few hours after exposure to pyrene in a coal tar-based shampoo or following dermal contact with either creosote or pyrene. The general public may also be exposed via dermal or inhalation routes to PAHs or from accidental ingestion of contaminated dust particles from the use of coal tar-based driveway sealants (Van Metre and Mahler 2010; Van Metre et al. 2012; Williams et al. 2012, 2013).

Occupational exposure to PAHs and other constituents of creosote may occur in several industries where workers are exposed to coal tar creosote, coal tar, coal tar pitch volatiles, or products containing creosote. Such occupations include jobs in the wood preserving industry, railroad work (installation and removal of crossties), treated lumber installation work involving structures such as fences or bridges, electric utility work involving treated poles, coke oven work, jobs in the rubber industry or tire plants, road paving work, roofing work, chimney cleaning, aluminum smelting work, iron foundry work, steel plant work, and site remediation work involving creosote-contaminated environmental media.

Individuals working in the wood-preserving industry comprise the largest portion of the population potentially exposed to coal tar creosote. Workers employed at creosote pressure-treatment facilities may be exposed by direct dermal contact or by inhalation of volatilized components. The IPCS CICAD on coal tar creosote summarized levels of components of coal tar creosote in the workplace air of several wood treatment facilities located in the United States, Netherlands, Finland, Sweden, and Germany (IPCS 2004). Potential exposure to coal tar creosote in these plants is minimized by using closed systems for receiving, transferring, mixing, storing, and applying the mixture to wood products. Similarly, dermal exposure from the handling of freshly treated wood is minimized by using highly mechanized processes.

ATSDR performed public health assessment and health consultation activities at four creosote wood treatment sites (ATSDR 2006, 2013, 2016b, 2020). The evaluations identified no public health hazards from exposure to creosote-related contamination in environmental media at the four sites.

Exposure of individuals installing treated fence posts, lumber, and timbers via inhalation of creosote volatiles (e.g., acenaphthene and naphthalene) can also occur when freshly treated materials are handled under calm, hot, sunny conditions (USDA 1980). Exposure may be greater during warmer months when ambient temperatures are higher. Acenaphthene, fluorene, phenanthrene, anthracene, and fluoranthene were observed to undergo more volatilization from creosote-treated wood at 30°C than at 4°C (Gevao and Jones 1998).

No data from the National Health and Nutrition Examination Survey (NHANES) on blood and urine levels of the entire creosote mixture are available. NHANES does provide data on blood and urine levels of some components of creosote mixtures, including several naphthalene and other polycyclic hydrocarbons (CDC 2024).

5.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Individuals living in the vicinity of hazardous waste sites and abandoned wood-treatment plants contaminated with coal tar creosote may experience higher levels of exposure than the rest of the general population. These environmental exposures generally are at a lower dose but of longer duration than the occupational exposures. Williams et al. (2012) noted that non-dietary intake of PAHs might be considerably higher for children who live in homes that use coal tar-based driveway sealants since young children often put their hands and objects into their mouths, which could include dust containing PAHs from these products.

Occupational exposure to PAHs from coal tar, coal tar creosote, and coal tar pitch are expected to be higher than for the general population. Assennato et al. (2004) reported that PAH concentrations in breathing zone air of coke-oven workers employed at a plant in Taranto, Italy ranged from 20.40 to 76.68 mg/m^3, with a median of 30.00 mg/m^3. The urinary levels of 1-hydroxypyrene samples ranged from 0.01 to 1.32 μmol/mol, with a median of 0.33 μmol/mol in the pre-shift, and from 0.01 to 31.04 μmol/mol, with a median of value of 2.41 μmol/mol, in post-shift. For workers exposed to creosote by chiseling a coal tar pitch layer by hand (one worker) or by handling creosote-impregnated wood (four workers), exposure to total PAHs and 4–6-ring PAHs was 50 times higher for the one worker exposed to the coal tar pitch layer while exposure to volatile naphthalene was >6 times higher for the wood handlers (Heikkilä et al. 1995). Total PAHs and 4–6-ring PAHs were measured at 440 and 290 μg/m^3, respectively, in the work area of the chiseler. Urinary concentrations of 1-hydroxypyrene were 2–4 times higher for the chiseler compared with the wood handlers. Volatile naphthalene was measured at 1,000 μg/m^3 in the work area of the wood handlers and 160 μg/m^3 in the work area of the chiseler. Urinary concentrations of 1-naphthol were 15–20 times higher for the wood handlers as compared with the chiseler.

Workers in a creosote railroad tie impregnation plant exposed to 1.5 mg/m^3 naphthalene, 5.9 μg/m^3 particulate PAH, and 1.4 μg/m^3 4–6-ring PAHs were measured for the urinary biomarker 1-naphthol (Heikkilä et al. 1997). The mean post-shift urinary concentration was 20.5 μmol/L; urinary concentrations in occupationally nonexposed male smokers were below the detection limit of 0.07 μmol/L. The study authors concluded that 1-naphthol was a good biomarker for determining exposure to volatile naphthalene from creosote but was not a good indicator of inhalation or dermal exposure to PAHs from creosote.

Jongeneelen (1992) related urinary concentrations of 1-hydroxypyrene for coke oven workers exposed to fumes containing PAHs to measured levels of coal tar pitch volatiles. This was done to equate the biological indicator data with lung cancer relative risk levels determined using epidemiological data obtained from U.S. and European coke plants. A urinary concentration of 2.3 μmol 1-hydroxypyrene/mol creatinine was equated with the American Conference of Governmental Industrial Hygienists (ACGIH) threshold limit value (TLV) of 0.2 mg/m^3 for coal tar pitch volatiles, and consequently with the relative risk for lung cancer of approximately 1.3 for a group of exposed workers. However, because the carcinogenic PAH fraction and the routes of exposure will also vary, the health risks related to exposure to coal tar creosote versus coal tar pitch volatiles versus coal tar will differ between exposed groups such as creosote and coke oven workers (Viau et al. 1995).

Coal-handling workers at a coke oven who were exposed to coal-tar sludge (67% coal tar) through dermal contact had increased urinary 1-hydroxypyrene concentrations following work shifts (Malkin et al. 1996). Urinary concentrations of the biomarker increased from a pre-shift mean of 1.00 μmol/mol creatinine to a post-shift level of 1.7 μmol/mol creatinine. The increases were attributed to dermal exposure, as exposure to volatile pyrene was determined to be minimal.

A review paper of studies using the concentration in urine of 1-hydroxypyrene as a biomarker of PAH exposure included levels reported in various studies (Strickland et al. 1996). The respective pre- and post-shift urinary excretion levels of 1-hydroxypyrene for coke oven workers were 0.89 and 2.47 μmol/mol creatinine; for asphalt pavers, respective levels were 1.35 and 1.76 μmol/mol creatinine.

Analysis of 319 breathing zone air samples and 31 general air samples indicated that exposures to coal tar pitch volatiles of workers at all jobs in 10 coke facilities surveyed in 1966 exceeded the threshold limit of 0.2 mg/m^3 time-weighted average (TWA) (Fannick et al. 1972).

Exposure to coal tar pitch volatiles (CTPV) has also been reported in aluminum smelter workers in Quebec (Lavoué et al. 2007). Exposures to CTPV were assessed by use of a job-exposure matrix (JEM) and estimated benzene-soluble material and benzo[a]pyrene levels. The JEM incorporated job and time period, including 28,910 jobs, from seven facilities from 1916 to 1999. Estimated exposures were 0.01–

68.08 μg/m^3 benzo[a]pyrene and 0.01–3.64 mg/m^3 benzene-soluble material. The exposures were lowest before 1940 and after 1980.

Air samples and urinary 1-hydroxypyrene in post-shift urine samples and next-day urine samples were analyzed for 36 creosote-exposed wood treatment plant workers (Borak et al. 2002). The results indicate that creosote is absorbed from both inhalation and dermal exposure, but that dermal absorption may be the predominant pathway. Compared to workers who showered following their shift, urinary 1-hydroxypyrene continued to increase in unshowered workers. Determination of volatized PAHs in the breathing zone was more useful than the traditional analysis of benzene soluble fraction of air samples for assessing creosote exposure.

Rubber processing workers at a tire plant in Poland who were occupationally exposed to coal tar pitch volatiles were found to have been exposed to excessive (>0.2 mg/m^3) levels of PAHs, including benzo[a]pyrene (Rogaczewska and Ligocka 1994). Measurements of benzo[a]pyrene were generally in the range of <4–142 ng/m^3, but were as high as 3,470–6,060 ng/m^3 for workers who weighed the raw materials.

In an investigation of the effect of decreased dermal exposure to creosote on the internal dose of PAHs in workers at a creosote wood impregnation plant, the use of Tyvek coveralls worn beneath outer work-clothes decreased the internal dose of pyrene (Van Rooij et al. 1993b). Workers not wearing the coveralls had total pyrene skin contamination of 47–1,510 μg/day and had urinary levels of 1-hydroxypyrene of 6.6 μg. For dermally protected workers, dermal pyrene contamination was approximately 35% less than that of the unprotected workers and urinary levels of 1-hydroxypyrene were 3.2 μg. The low level of efficacy was attributed to uncovered skin areas (face, wrists, ankles). Volatile pyrene in the breathing-zone air was measured at 0.3–3.0 μg/m^3. The study authors determined that for creosote workers, the level of dermal exposure to PAHs is the main determinant of the internal exposure dose; 15 times more pyrene was absorbed through dermal uptake than through respiratory uptake. Data from earlier studies indicate that the daily skin contamination with pyrene was higher for creosote workers (median of 350 μg) compared with that measured for coke oven workers (70 μg) and road pavers (117 μg); for aluminum workers, a pyrene level of 395 μg was measured (Van Rooij et al. 1993b).

Note that susceptibility of children and other sensitive populations is discussed in Section 3.2.