CREOSOTE - ENVIRONMENTAL FATE

I. Executive Summary/Creosote Use Overview

Creosote is an oil-based wood preservative that is used primarily for preserving wood used in railroad ties and utility poles. Coal tar creosote (i.e., the P1/P13 and P2 fractions of creosote) are obtained from the carbonization of bituminous coal and are mixtures of about 200-250 substances consisting of simple polyaromatic hydrocarbons (PAH), multi-aromatic fused rings, cyclic nitrogen-containing heteronuclear compounds, and phenolic substances. It is estimated that about 85 percent of the creosote mixture consists of PAHs and about 15 percent cyclic heteronuclear nitrogen- and oxygen-containing molecules. The Agency made the decision to base the environmental fate studies on the PAH constituents only because they constitute the majority of the percent mass of the P1/P13 and P2 fractions of creosote.

Coal tar creosote has been used as a wood preservative pesticide for over 125 years. According to the American Wood Preservers' Association (AWPA,1997) statistics, an estimated 728 million cubic feet of wood was treated with preservatives, and creosote and its mixtures represent 13.3 percent of the treatments. Nearly all railroad crossties, switch ties and bridge timbers are pressure-treated with creosote, as well as 14.6 percent of utility poles (Webb, n.d.).

Process wastewater, dumpsite leachate, storage tank leaks, and spills are the major sources of creosote releases to the environment (Merril and Wade, 1985). The environmental fate of creosote in this report focuses primarily on the exposure of PAHs into three environmental compartments: 1) leaching into surface and ground waters from the railroad ties and utility poles; 2) migration into soil/sediments from the railroad ties and utility poles, and 3) bioaccumulation in the aqueous and benthic organisms.

<u>Environmental Fate and Transport</u>

The major route of exposure from creosote is through water and soil, and subsequent transfer from these environmental compartments into the aquatic and benthic organisms (bioaccumulation).

a. Abiotic Degradation

PAHs are fused aromatic polycyclic rings that have no hydrolyzable hydrogens. The solubility of these compounds is very low in water. Environmentally, hydrolysis does not appear to be an important pathway for dissipation of the composite mixture of PAHs in water. However, some PAH molecules persist for long periods of time in water. On the other hand, photooxidation commonly occurs with creosote PAHs. The photolytic half-lives of the PAHs in aqueous medium depend on the season, geographical location, surface water measurements, and complexity of the parent molecules. In most cases, the photolytic half-lives of PAHs are short.

b. Mobility

Once introduced to an aquatic environment, creosote components are subject to several fractionation processes. Many PAHs adsorb to sediments and may persist in the environment for long periods of time. Eventually, sediment adsorbed PAHs may dissolve or become resuspended in water by tides, storms, bioturbation, shipping, or dredging. Additionally, colloidal matter present in a cresote-contaminated environment has been found to affect the desorption rate of specific PAHs. PAHs from creosote-treated utility poles and/or railway ties tend to leach initially and remain in the sediment surrounding the poles or railroad ties, not migrating far from the wood.

c. Biodegradation

Most PAHs have a tendency to biodegrade under aerobic conditions. A number of aerobic soil metabolism studies on PAHs conducted at various contaminated sites as well as in simulated microcosms reported that low molecular weight PAHs generally metabolized in aerobic conditions and the greater the available oxygen, the higher the biodegradation level.

d. Bioaccumulation in Fish

The major components of the PAHs in creosote have shown the ability to form neutral and oxidized transformation products under aerobic soil/aquatic conditions. Numerous studies have shown that photooxidized transformation products of these PAHs are bioaccumulative and result in adverse effects on the aqueous biota as well as on the organisms in the soils and benthic sediments. In aquatic habitats, fish, shellfish, and crustaceans readily bioaccumulate PAHs from the environment and store these at high levels in the tissues. Some PAHs, particularly those that have a high molecular mass have a higher tendency to adsorb to soil organic carbon.

Data Summary

The following discussion briefly outlines the measured and/or estimated environmental fate and transport data. Greater detail is presented in the discussions in Section II.

a. Volatility

In a study by Lindhardt and Christensen (1996), coal tar contaminated soil, below a 5 cm deep layer of uncontaminated soil, was monitored for 53 days. Where microbial activity was inhibited, some PAH fluxes stabilized after 10 to 20 days. Other PAHs were measurable only when using a soil cover with low organic content. When adapted for degradation of naphthalene, some PAH fluxes approached the detection limit at 5 to 8 days. In a volatilization study reported by Gevao and Jones (1998) for five PAHs in treated wood, mean PAH values ranged from 2.57 ± 1.52 mg/m^2 treated wood/day and 29.5 ± 6.1 mg/m^2 treated wood/day at 4 ºC and 30 ºC, respectively. The half-life ranged from 0.70 to 31 years at 4ºC, and from 0.3 year to 1 year at 30 ºC. A long-term study conducted at 4 ºC, showed that the volatilization rate remained constant for about seven weeks, after which, more than 85 percent of the PAHs remained.

b. Photolysis

A photolysis study in natural sunlight in aqueous (~5 x 10^{-8} M) media with selected commercially available highly pure PAHs was conducted by Kirso (1991). The photooxidation half-lives of these PAHs ranged from 3.56 to 374.94 minutes in water and from 0.68 to 190.15 minutes in benzene. The study also collected data on photolysis by natural sunlight of benzo[a]pyrene under open-sea conditions at northern and southern latitudes. First-order rate constants ranged from $0.7x10^{-4}$ to $4.2x10^{-4}$ sec^{-1}, and half-lives ranged from 27.5 to 146.2 minutes. Half-lives have also been reported in the international scientific literature, under a variety of experimental conditions, for most of the PAHs present in the P1/P13 and P2 fractions of creosote. These half-lives range from 0.5 hours for anthracene (alumina atmospheric particulate substrate) to 410 Days for 2-Methyl naphthalene (summer sunlight in surface water).

c. Aerobic/anaerobic Metabolism

Bench and pilot scale studies (Mueller, et al., 1993) have demonstrated the ability of a sequential inoculation process, using selected microorganisms, to enhance the bioremediation technologies for the treatment of groundwater contaminated with creosote. For PAHs, biodegradation values were determined to be as high as 98.0 percent. USGS (Godsy, et al., 1992) analyzed groundwater near a contaminated site that was anaerobic with the presence of methane and hydrogen sulfide (indicating the presence of microbes). The concentrations of PAHs generally decreased with distance from the source of contamination. A similar study at the site (Goerlitz, et al., 1985) showed that the level of naphthalene progressively decreased with increasing well depth.

Mineralization of some PAHs in intertidal sediments showed dependence on the amounts of the polyaromatics present, oxygen level, and pre-exposure time (Bauer and Capone, 1985). A similar study (Bauer and Capone, 1988) reported that the pre-exposure of some PAHs to benzene and other PAHs under aerobic conditions in shallow marine sediments accelerated their mineralization. Bouwer, et al. (1996) reported that the bacteria present in contaminated aquifer sediments were able to degrade low molecular weight PAHs, and that greater biodegradation occurs with higher oxygen concentrations. Another study using bacterial cultures obtained from washing creosote-contaminated soils (Chapman, et al., 1995) showed that approximately 72 percent of the low molecular weight PAHs were easily degraded by bacterial enrichments.

In a study on anaerobic soil biodegradation (Genther, et al., 1997), contaminated soil samples were collected from sediments in a creek bed and batches of an artificial PAH mixture were prepared and inoculated with the contaminated soil samples under various conditions. The majority of the PAHs did not degrade in the samples collected at 5 to 8 cm deep. For the 12-meter soil samples, no appreciable biodegradation processes were observed for most of the PAHs. In a study reported by Hurst, et al., (1996), contaminated soil samples were spiked with ^{14}C-pyrene and seventeen PAHs, followed by oxygen as soil gas. After 70 days, 45 to 55 percent of the ^{14}C-pyrene was mineralized in 2 and 21 percent oxygen. For eight of the non-radiolabeled PAHs, biodegradation in an oxygen atmosphere ranged from 6.2 to 57 percent. Analysis of creosote-contaminated soil and groundwater (Mueller, et al., 1991) showed that bicyclic PAHs and phenolics are metabolized readily by the microorganisms in the aquatic soil while the PAHs with a higher number of fused rings biodegrade more slowly.

d. Leaching and Adsorption/Desorption

The results of a two-step fractionation procedure (Villholth, 1999), showed that the PAHs partitioned to the course (i.e., >100 nm) colloid fraction (log K_{oc}) and was linearly correlated with the PAH octanol-water partition coefficient (log K_{ow}). Rutherford, et al. (1997) reported that the desorptive partition coefficient (K_d) for nonbioremediated soil at one location was significantly greater than the nonbioremediated soil at another location, from the same contaminated site. However, after bioremediation, no significant difference was found between the two soils. A laboratory study (Priddle and MacQuarrie, 1994) was conducted to evaluate the dissolution of industrial creosote in water. Starting with a 10 percent creosote effluent, the creosote components reached equilibrium with the aqueous phase in about 60 hours. The concentrations of the PAHs decreased steadily throughout the monitoring period. A study (Padma, et al., 1999), showed that sediments contain more intermediate and high molecular weight PAHs, in contrast to the water-soluble fraction, which contains high levels of low molecular weight compounds.

The Electric Power Research Institute (EPRI , 1992) tested samples from poles and crossarms, from across the United States using TCLP. The maximum total cresol concentration (all three isomers) present in the leachates was 14.95 mg/mL, and the mean concentration was 1.63 mg/mL. Wendt, (1996) performed a study on docks originally treated with creosote. Mean concentrations of 12 PAHs in sediment were 978.3 and 690.0 µg/kg (dry wt.), for the <1 m and >10 m samples, respectively. Mean PAH concentrations in oysters were 3547.3 and 2057.6 µg/kg (dry wt.) for the <1 m and >10 m samples, respectively. An Agency sponsored study (Middaugh, et al., 1991) was conducted on the leaching of creosote components from an abandoned site to a freshwater stream that flows into a bay. The maximum PAH concentration in the groundwater was reported for phenanthrene as 32.8 mg/kg. In a study reported by Wan (1994), runoff water from pressure-treated utility and telephone wood poles was collected from utility and railway ditches, as well as ditches of parklands, farmlands, and railway ROWs. PAHs were not detected in the parkland ditches; however, they were found in farmlands and in utility and railway ROW ditches.

Pressure-treated wood blocks were tested in a terrestrial microcosm chamber (Gile, et al., 1982). After 2.5 months, 95 percent of the pesticides remained in the wood and most of the materials that leached remained in the upper soil layer immediately surrounding the pine blocks. PAHs were also detected in the plants of the microcosm, in the invertebrates and in the body of a gravid gray-tailed vole. In a field study (Bestari, et al., 1998a), the fate of creosote components in aqueous medium, sediment, and adsorption processes were studied using microcosms. The concentrations of PAHs showed a rapid decline in the water. In sediments, total PAHs declined after day 28. A similar declining trend was shown by the PAHs on PVC strips. A mass balance calculation reported a loss of 88.3 percent of PAHs from the microcosms after one month. The half-life of most PAHs in water was reported to be about one week. In another study by Bestari, et al., (1998b), creosote pressure-treated wood was examined in a similar microcosm. The analysis showed that the concentration of PAHs increased rapidly through Day 7 after the treatment and then decreased to close to the background by the end of the study.

e. *Bioaccumulation*

Stegman and Teal (1973) and Varsani, et al. (1978) showed that in an aquatic habitat, organisms readily accumulate PAHs from the environment and store them at a high level in their tissues. A study reported by Southworth, et al. (1978) showed that bioaccumulation corresponded to the octanol/water partition coefficients (K_{OW}). A bioaccumulation study on clams was conducted after a creosote spill. Results showed that the levels of PAHs increased gradually for two weeks and increased dramatically by week four at the monitoring station closest to the spill site. The control station in the study showed evidence of depuration after two weeks and equilibration after four weeks. Other studies (Politzer, 1985; Neff, 1976) also showed that depuration of PAHs in bivalves vary from a couple of weeks to a few months. Howard, et al. (1991) report that the half-lives of PAHs increase as the complexity of the molecules increase, and K_{ow} values show a similar trend. The Log K_{ow} increases as the complexity of the molecule increases. In general, the half-lives in air and water are lower than in soils/sediments. Molecules with a longer half-life also exhibit persistence. PAHs show more persistence in soils/sediments.

A study using Bluegill sunfish reported the uptake half-life for anthracene and benzo[a]pyrene as 0.019 hours (Spacie, et al., 1983). Biotransformation for anthracene and benzo[a]pyrene were reported as 0.22 nmol/hr and from 0.044 nmol/hr to 0.088 nmol/hr, respectively. Depuration half-lives for anthracene and benzo[a]pyrene were 17 and 67 hours, respectively. Bioconcentrations for both PAHs were lower than predicted from the K_{ow}. Bioaccumulation of PAHs were investigated on the bivalve mollusk (*Macoma balthica*). Tay, et al., (1992) suggest that the duration of 30 days may have been too short for uptake of the PAHs into the *Macoma* system. The high organic content in the sediment that may have prevented the bioaccumulation process.

PAHs were found in mollusk species studied (Elder and Dresler, 1988) at a bay, 500 m from a creosote wood-preserving site. The study also found that the sediments in the drainage streams were heavily contaminated with PAHs. Analysis of sediments at these sites showed very little PAH contamination except close to the wood-preserving facility. The study also reported that the bioaccumulation of 3 PAHs in both species of mollusks was ten times higher at the test site than at the control site. Most of the PAHs were insoluble in water and the solubilities were inversely related to the molecular weights of the polyaromatics. In general, molecules with lower molar masses had a tendency to bioaccumulate more than high molar mass substances. Bioaccumulation was also dependent on the concentrations of a substance. Bruner, et al. (1994) reported that pre-spawning mussels had greater bioconcentration factors and a faster rate of accumulation for benzo[a]pyrene than post-spawning mussels. Lipid content, however, did not influence bioconcentration factors, rates of accumulation, or rates of depuration. Accumulated PAH values in the muscles of selected marine mammals, on a dry weight basis, were reports by Hellou, et al. (1990) as 0.10 to 1.21 ppm chrysene-equivalents.

f. Migration of PAHs From Poles to Soils

Concentrations of PAHs varied from 25 to 50 0g/g in 56 soil samples collected radially around 14 creosote-treated poles. Some PAHs were present at low concentrations, and some were persistent in the soil around the poles. The concentration of all the components decreased as the distance from the poles increased (Mississippi State University, 1981). McGroddy and Farrington (no date) showed that the PAH concentrations measured in sediments and porewaters from three cores were notably lower than the amounts of the PAHs predicted by two- and three-phase equilibrium partitioning models.

II. Environmental Fate and Chemistry of Creosote

A. Chemical Profile

Common Name(s):	Creosote, Creosote Oil, Dead Oil, Brick Oil, Coal Tar Oil, Creosote P1, Heavy Oil, Liquid Pitch Oil, Wash Oil, Creosotum, Cresylic Creosote, Naphthalene Oil, Tar Oil, AWPA #1, and Preserv-o-sote
Chemical Name:	Coal Tar Creosote
Trade Name:	Sakresote 100
Formulations:	Distillate mixture obtained from bituminous coal; oil-based

Physical/Chemical Properties:

Molecular formula:	Not applicable
Molecular weight:	Not applicable
Physical State:	Translucent brown to black; yellowish to dark green-brown; oily liquid
Melting Point:	Not available
Boiling Point:	194-400°C
Viscosity:	14.60 mm/s (P1/P13); 15.5 mm/s (P2)
Vapor Pressure:	11.1 mm Hg at 24.4°C (P1/P13); 8.6 mm Hg at 24.4 to 24.5°C (P2)
Dissociation Constant:	3.247 (pKa, P1/P13); 3.311 (pKa, P2)
Solubility:	313 µg/mL (P1/P13); 306 µg/mL (P2); temperature not specified
Henry's Law Constant:	Not available
Octanol/Water Partition Coefficient (log K_{ow}):	1.0 (pH not specified)

B. Environmental Fate Assessment:

This section presents information on the environmental fate and transport of creosote, followed by summaries of the articles that contan the supporting data and information.

1. Environmental Fate and Transport of Creosote

The major uses of creosote since 1988 have been railroad ties, crossbars, decks on marinas and utility poles. Polyaromatic hydrocarbons (PAHs) constitute the highest percent (85%) of coal tar creosote while the phenolic substances are about 10 percent, and N– and S-containing substances represent the remainder of the mixture. Most of the PAHs are non-volatile, therefore; creosote normally does not contaminate the air. The major route of exposure from creosote is through water and soil, and from these environmental compartments into the aquatic and benthic organisms (bioaccumulation).

Abiotic Degradation

The PAHs are fused aromatic polycyclic rings which have no hydrolyzable hydrogens and the solubilities of these compounds are very low in water. Environmentally, hydrolysis does not appear to be an important pathway for dissipation of the composite mixture of PAHs in water; however, some molecules like benzo[k]fluoranthene and benz[a]pyrene could persist in water.

Very few studies are reported in the open literature on field volatilities for PAHs present in creosote. Gevao et al. (1998) showed that acenaphthene, fluorene, phenanthrene, anthracene, and fluoranthene volatilized at a faster rate at 30°C than at 4°C. The study also showed that 85 percent of these components remained in the wood after seven weeks. The rate volatilization was slow for acenaphthene (half-life, one year) and fluoranthene (half-life, one year). In most cases, the initial rates of dissipation are caused by partitioning between the wood and air and biodegradation in the presence of microbial populations. Therefore, exposure to air does not appear to be an important factor in fate assessment for most PAHs.

Photooxidation is a common phenomenon for the creosote PAHs. The photolytic half-lives of the PAHs in aqueous medium are dependent on the season, geographical location, surface water measurements, and complexities of the parent molecules (two fused rings vs. five fused rings). In most cases, half-lives under the conditions mentioned do not appear very long. Because of this, and the fact that most of the PAHs are not readily soluble (except for a few low molecular weight ones), the PAHs may not be a problem in surface and groundwater runoffs. However, it should be noted that the photooxidized products of PAHs are stable; therefore, may persist in air/water and soils and become an environmental concern as these photooxidized products are also bioaccumulative.

Mobility

Once introduced to an aquatic environment, creosote components are subjected to several fractionation processes. Many PAHs adsorb to sediments and may persist for long periods of time. Creosote contaminated sediments usually contain relatively higher levels of hydrophobic PAHs than whole creosote (Bieri et al., 1986). Eventually, sediment adsorbed PAHs may dissolve or become resuspended in the water by tides, storms, bioturbation, shipping, or dredging. As a result, local biota may be exposed to low level PAHs over the long term (Fowler et al., 1993). Therefore, the adsorption/desorption processes in water involved with creosote-derived PAHs are a significant consideration in fate assessments of creosote contamination.

Additionally, colloidal matter present in a cresote-contaminated environment has been found to affect the desorption rates of specific PAHs. One study found that PAHs partitioned to course (>100 nm) colloid fractions and were linearly correlated with the PAH octanol-water partition coefficient, indicating the partitioning was hydrophobic (Villholth, 1999).

The PAHs from creosote-treated utility poles and/or railway ties tend to leach out initially in the first seven days and remain in the sediment surrounding the poles or railroad ties not migrating far from the wood. Most of the PAHs, however, tend to stay inside the wood (~85%). One study showed that background levels for PAHs leached from wood were attained within three months and may have been due to photolysis or biodegradation of the PAHs. A detailed study of 200 U.S. estuaries showed that PAHs that leached from the treated wood of decks and bulkheads, 175 had muddy sediments. Higher amounts of PAHs leached into such soil types. No systematic work has been carried out on all the PAHs and the representative soil types to show which one would have a higher tendency of retention for the PAHs.

Migration studies of PAHs into groundwater have shown that migration of some of the PAHs have been observed. Vertical or lateral migration of the PAHs from the utility poles indicated that at ground level the migration was not significant beyond 150 meters from the site of contamination (base of utility poles). The vertical or downward migration of the PAHs was even more limited and the existence of the PAHs were not found below a 12 meter depth.

Biodegradation

Most of the PAHs have a tendency to biodegrade under aerobic conditions. It has been reported that over eighty percent of biodegradation occurs in the first month after the wood preservative application, with the exception of benz[a]pyrene and benzo[k]fluoranthene, which have shown resistance to biodegradation. A number of aerobic soil metabolism studies on PAHs conducted at various contaminated sites as well as in simulated microcosms reported that low molecular weight PAHs generally metabolized in aerobic conditions and the greater the oxic environment, the higher the biodegradation level.

Aerobic degradation of PAHs associated with soil and groundwater often leads to a rapid depletion of dissolved oxygen which eventually decreases the redox potential. This decrease in the redox potential can result in favorable environments for denitrifying, sulfate-reducing, or methanogenic microbes. Therefore, anaerobic transformations may be a significant factor in oxygen-depleted habitats (Karthikeyan and Bhandari, 2001). Under these conditions, anoxic or anaerobic degradation stimulated by denitrifying or sulfate-reducing bacteria can become an important pathway for the cleanup of contaminated sites.

Bioaccumulation in Fish

The major components of the PAHs in creosote have shown the ability to form neutral to oxidized transformation products under aerobic soil/aquatic conditions. For example, fluorene forms hydroxy fluorene and acenaphthene converts into diacetic acid acenaphthene. These oxidized species are stable and bioaccumulative. Numerous studies have shown that photooxidized transformation products of these PAHs are bioaccumulative and result in adverse effects on the aqueous biota as well as on the organisms in the soils and benthic sediments.

In aquatic habitats, fish, shellfish, and crustaceans readily bioaccumulate PAHs from the environment and store these at high levels in the tissues. Seven PAHs: naphthalene, anthracene, phenanthrene, pyrene, 9-methyl anthracene, benz[a]anthracene and perylene were shown to bioaccumulate in *Daphnia pulex*. PAHs like naphthalene, biphenyl/acenaphthylene, fluorene, phenanthrene/anthracene/chrysene, and benzopyrene were found to bioaccumulate in clams (*Rangia cuneate)*. The most dramatic increases were in cases such as benz[a]anthracene/chrysene which reported a bioaccumulation of 41 ppb (week zero) and increased to 190 ppb (week 4). Similarly, benzopyrene bioaccumulated from 8 ppb (week zero) to 600 ppb (week 4). Depuration was within two weeks. This study was conducted after the creosote spill into the Bayou Bonfuca at the American Creosote Works Plant Site at Slidell, Louisiana.

A study on benthic invertebrates showed a bioaccumulation concentration ranging from 0.10 to 11.00 ppm. A bioconcentration study on zebra mussels in the Great Lakes found that pre-spawning species (high lipid) bioaccumulated benzo[a]pyrene at a faster rate than the post-spawning (low lipid) species.

Bioaccumulation data on marine mammals are not readily available, and only one study on whales and seals has been reported. That study indicated a bioaccumulation of 0.10 and 1.21 ppm in the muscles of these mammals, respectively.

Some of the PAHs, particularly those that have a high molecular mass (higher number of the fused aromatic rings) have a higher tendency to adsorb to soil organic carbon. Such adsorption coefficients (K_{OC}) have been reported in literature. Some PAHs with a high K_{oc} can bind strongly with the organic carbon of the soils/sediments and may not be bioavailable to the aquatic organisms. However, if the octanol/ water coefficients (K_{ow}) is high, and if the PAHs are desorbed from the soils/sediments to which they are bound, some of these PAHs can become bioaccumulative to the benthic organisms.

Recently, it was suggested that based on theoretical calculations and modeling, the half-lives of the PAHs obtained from coal tar creosote can be estimated in air, water, soils and sediments. From these calculations and modeling, one can arbitrarily divide the PAHs into 3 distinct groups: PAHs with two fused rings, PAHs with three fused rings and PAHs that have 4 to 5 fused rings. The half-lives in the environmental compartments (air, water, and soils) for PAHs are as follows: the half-lives of two fused rings PAHs < three fused rings < 4/5 fused rings. The K_{ow} values lie between 3 and 4 for PAHs with two fused rings, between 4 and 5 for PAHs with 3 fused rings, and at 6 or above for the third group of the PAHs. In general, the half-lives in air and water environmental compartments are much lower than in soils/sediments because the soil adsorption coefficients are higher. The longer the half-life, the greater the persistence of the PAHs in soils.

Some of the 4/5 fused ring PAHs are more persistent in soils and sediments and since their K_{ow} values are higher, they can bioaccumulate because some of them adsorb onto soils and they may not be bioavailable for benthic organisms.

The third group of PAHs show a higher degree of bioaccumulation, persistence in soils and water, resistance to biodegradation and photooxidation. Additionally the have less of a tendency to leach from the wood structure and have high sorption constants to soils. On the other hand, these higher members of the PAHs (4/5 fused ring compounds) are readily soluble in water and their percent on a mass basis in the creosote mixture is very low compared to the first group (2 fused rings) of the PAHs, and may not be available for biomagnification and migration into surface and ground waters.

2. Supporting Data Summary

Registrant studies have not been submitted to the Agency and specific guideline requirements have not been fulfilled for creosote. Therefore, the Agency has carried out an open literature search for environmental fate studies on creosote mixtures and has decided to make the assessment based on the PAH constituents only. The literature search on environmental fate studies provided data on the volatility, photolysis in water, aerobic/anaerobic metabolism, leaching and adsorption/desorption, bioaccumulation in aquatic and benthic organisms, and migration from poles into soils. The following are summaries of studies obtained from the literature search.

Volatility

Lindhardt, B. and T.H. Christensen, 1996
A laboratory volatilization study was conducted on the non-steady-state fluxes of aromatic hydrocarbons from coal tar contaminated soil. The contaminated soil samples, obtained from Holte, Denmark, were placed below a 5 cm deep layer of uncontaminated soil and monitored for 53 days. The contaminated soil contained 50 to 840 $\mu g/cm^3$ of 11 selected aromatic hydrocarbons. In analyses where the microbial activity was inhibited, the fluxes stabilized on a semi-steady state level for monocyclic aromatic hydrocarbons, naphthalene and 1-methylnaphthalene after 10 to 20 days. Acenaphthene and fluorene fluxes were measurable only in experiments using a soil cover with a low organic content. When the soil cover was adapted to degrade naphthalene, the fluxes of naphthalene and 1-methylnaphthalene were approaching the detection limit at 5 to 8 days.

Gevao, B. and K.C. Jones, 1998
A volatilization study for five PAHs (acenaphthene, fluorene, phenanthrene, anthracene and fluoranthene) from treated (painted) wood was conducted in the United Kingdom. This laboratory study was performed using glass chambers equipped with an air inlet/outlet in 4°C and 30°C environments. Wood samples painted with 110 grams of creosote were placed in the glass chambers and the air traps were changed at each sampling interval. Samples were stored for 2 to 4 weeks at -17°C prior to extraction. The rate of desorption was analyzed using first order kinetics for all five PAHs and was found to be higher at 30º C than at 4ºC. The mean PAH values ranged from 2.57 ± 1.52 mg/m^2 treated wood/day and 29.5 ± 6.1 mg/m^2 treated wood/day at 4°C and 30°C, respectively. From the desorption rates, the half-life at 4ºC ranged from 0.70 year to 31 years for fluoranthene and acenaphthene, respectively. When the temperature was

raised to 30°C, the half-life ranged from 0.3 year to 1 year for fluroanthene and acenaphthene, respectively. Following a long-term study at 4°C, it was observed that the volatilization rate was constant for about seven weeks after which it was estimated that >85 percent of the PAHs remained in the wood. The authors noted that initial desorption rates were caused by partitioning between the wood and air, and by the rates of compound diffusion from the interstices of the wood.

Photolysis

Kirso, U., 1991
A photolysis study in natural sunlight in aqueous (~5 x 10^{-8} M) PAH, aza-PAH and benzene media was performed with selected PAHs. Table 1 summarizes the photooxidation half-lives of these commercially available highly pure PAHs. The study also collected data on photolysis by natural sunlight of benzo[a]pyrene under open-sea conditions at northern and southern latitudes. In this European study, first-order photooxidation rates were measured and first-order rate constants were found. The Agency has calculated the half-lives using the first-order rate constants from this study. The results of the calculations are presented in Table 2.

Table 1. Photochemical Oxidation and Half Lives of Selected PAHs

Compound	Half-life in Water (minutes)	Half-life in Benzene (minutes)
Fluorene	119.97	2.05
Anthracene	8.06	*
Phenanthrene	64.95	107.05
Triphenylene	93.93	136.80
Pyrene	19.99	97.47
Chrysene	25.88	100.54
Benz[a]anthracene	20.92	41.72
Benz[b]anthracene	3.56	0.68
Dibenz[a,c]anthracene	25.57	55.06
Dibenz[a,h]anthracene	22.94	55.06
Dibenz[a,j]anthracene	16.58	41.04
Dibenzo[a]pyrene	15.5	34.20
Dibenzo[e]pyrene	22.94	117.99
Perylene	374.94	51.98
Coronene	312.01	63.95
Benzo[k]fluoranthene	110.98	171.00
Benzo[b]fluoranthene	312.01	190.15

Notes: 1. Not all the PAHs used in the study are present in the P1/P13 and P2 fractions of the creosote.
2. PAHs with high molar masses have higher photooxidation half-lives than smaller PAHs with small molar masses.
3. * = Could not be determined under the experimental conditions.

Table 2. Rate Constants and Half-lives of Photooxidation of Benzo[a]pyrene by Sunlight in Sea Water

Region	Water Temperature (^oC)	Initial Concentration (10^{-8} M)	Rate Constant K (10^{-4} sec.$^{-1}$)	Half-life (minutes)
Bering Sea	14	1.47	1.69	68.3
Bering Sea	16	4.20	1.60	72.2
Tropical Pacific	26	6.59	2.84	40.6
Tropical Pacific	27	2.06	1.60	72.2
Lagune Caroline Atoll	27	1.90	2.99	38.61
Lagune Caroline Atoll	27	0.70	4.20	27.50
Baltic Sea	6	15.6	0.70	146.2

Notes: 1. 'K' is a first-order rate constant.
2. The Rate constants of photooxidation of benzo[a]pyrene (and half lives) are higher in tropical longitudes than in northern and temperate climatical zones.

Experimental and calculated data on the photolysis of most of the PAHs present in the P1/P13 and P2 fractions of creosote can be found in the international scientific literature. These data are summarized in the Table 3 below.

Table 3. Photolysis of Selected PAHs Found in P1/P13 and P2 Fractions of Coal Tar Creosote

Compound	Description	Half lives
Naphthalene	Direct sunlight, 40°N, midday, midsummer (calculated)	71 Hours (Harris, 1982)
	Distilled water at 25 º C	25 Hours (Fukuda, 1988)
Quinoline	Sunlight at 40°N, aqueous hydrolysis, pH 6.9	3851 Hours, summer 535 Hours, winter (Mill et al., 1981)
1-Methyl naphthalene	Summer sunlight in surface water	180 Days (Miller, 1985)
2-Methyl naphthalene	Summer sunlight in surface water	410 Days (Mill et al., 1981)
	Distilled water	16.4 Hours (Fukuda, 1988)
Acenaphthene	Determined by rotary photoreactor technique on different atmospheric particulate substrates: -- silica gel -- alumina -- fly ash	 2.0 Hours (Behymer & Hites, 1985) 2.2 Hours (Ibid.) 44 Hours (Ibid.)
	Distilled water(irradiated light at wavelength 290 nm)	3 Hours (Fukuda, 1988)
Fluorene	Determined by rotary photoreactor technique on different atmospheric particulate substrates: -- silica gel -- alumina -- fly ash	 110 Hours (Behymer & Hites, 1985) 62 Hours (Ibid.) 37 Hours (Ibid.)

Table 3. Photolysis of Selected PAHs Found in P1/P13 and P2 Fractions of Coal Tar Creosote (Con't)

Compound	Description	Half lives
Anthracene	Midsummer sunlight: -- deep, slow, somewhat turbid water -- deep, slow and muddy water -- deep, slow and clear water -- Shallow, fast and clear water -- shallow, very fast and clear water	 173.2 hours (Southworth, 1977) 693 hours (Ibid.) 38.5 hours (Ibid.) 8.1 hours (Ibid.) 2.91 hours
	At 35° on latitude -- summer -- winter	 1.6 hours (Ibid.) 4.8 hours (Ibid.)
	At 35° N, winter -- in water	 4.62 hours (Callahan, 1979)
	Different atmospheric particulate substrate: -- silica gel -- alumina -- fly ash -- distilled water	 2.9 hours (Beymer & Hites,1985) 0.50 hours (Ibid.) 48 hours (Ibid.) 1.0 hour (Fukuda, 1988)
Carbazole	40°N , midday, sunlight in late Jan. and river (calc.)	6.0 hours (Smith, 1978)
	In eutrophic pond and eutrophic lake	15.0 hours (Ibid.)
	In oligotrophic lake	3.0 hours (Ibid.)
	Aqueous medium	1.0 hours (Ibid.)
Fluoranthene	Atmospheric/aqueous photolysis, based on the measured sunlight photolysis rate constant in water -- summer sunlight, surface water	21 hours (Howard, 1991) 160 days (Mabey, 1982)
	Different atmospheric particulate substrates: -- silica gel -- alumina -- fly ash	 74 hours (Behymer & Hites, 1985) 23 hours (Ibid.) 44 hours (Ibid.)

Table 3. Photolysis of Selected PAHs Found in P1/P13 and P2 Fractions of Coal Tar Creosote (Con't)

Compound	Description	Half lives
Chrysene	Different atmospheric particulate substrates: -- silica gel -- alumina -- fly ash	100 hours (Behymer & Hites, 1985) 78 hours (Ibid.) 38 hours (Ibid.)
Acenaphthylene	Different atmospheric particulate substrates: -- silica gel -- alumina -- fly ash	0.7 hours (Behymer & Hites, 1985) 2.2 hours (Ibid.) 44 hours (Ibid.)
Benz[a]anthracene	Aquatic	10-50 hours (Callahan, 1979)
	Stream	20 hours (Smith, 1978)
	Eutrophic pond or lake	50 hours (Ibid.)
	Oligotrophic lake	10 hours (Ibid.)
	Aquatics	0.58 hours (EPA Report 600/7-78-074)
	Early March	0.2 days (Zepp, 1980)
	-- 1% acetonitrile in filter-sterilized natural water at 313 nm wave length	5 hours
	Different atmospheric particulate substrates: -- silica gel -- alumina -- fly ash	4.0 hours (Behymer & Hites) 2.0 hours (Ibid.) 38 hours (Ibid.)

Aerobic/anaerobic Metabolism

Brenner R.C. et al. 2002

This study examined the site of the former Wyckoff wood-treatment facility, near Seattle, WA. The facility used large quantities of creosote in wood-treating operations from the early 1900s to 1988. Historical creosote seepage into Eagle Harbor resulted in PAH contamination in the sediment. Ten sediment cores were taken in the harbor, each divided into 5-cm segments. The cores taken adjacent to the former facility were dominated by a strong creosote signature in the deepest portions of the cores, below the 15 to 20 cm depth range. The mid-harbor cores were dominated by a strong urban runoff signature in the top 30 cm, corresponding to the present, and dating back to 1908. Below the 45 to 50 cm depth range corresponds to pre-1900. In these sediments, PAHs were dominated by a natural background hydrocarbon signature. Equally interesting to the urban runoff signature was the absence of any sign of creosote in these mid-harbor cores.

The cores exhibited varying degrees of weathering, identified as unweathered, slightly weathered, moderately weathered, and severely weathered. The severely weathered creosote sample contained a very low percentage of 2- and 3-ring PAHs but exhibited a sharp increase in the percent of fluoranthene and other higher molecular weight PAHs, compared to the unweathered creosote sample. Creosote weathering resulted in the relative loss of lower molecular weight PAHs. There was significant loss in the percentage of 2-ring PAHs; 100 percent in some cases. Some samples exhibited no loss of unweathered creosote, and 68 percent loss of slightly weathered creosote. The C_3 and C_4-naphthalenes were more resistant to weathering and tended to be reduced more slowly.

Selifonov, S.A. et al., 1998
This study examined the use of ^{13}C nuclear magnetic resonance (NMR) to assess the fate of [1-^{13}C] acenapthene in creosote polycyclic aromatic compound mixtures degraded by bacteria. Site-specific labeling at the benzylic position of acenapthene allows use of ^{13}C NMR spectroscopy to study the biodegradation of acenapthene by various bacterial cultures degrading aromatic hydrocarbons of creosote. An undefined mixed culture of bacteria (CREOMIX) was obtained from creosote-contaminated soil obtained from the American Creosote Works site in Pensacola, FL. Two individual strains, BR and BC, were isolated from the CREOMIX culture. Neither strain BR nor BC could grow on acenapthene as the sole carbon and energy source. But, acenapthene was oxidized by these strains when naphthalene was added as a cosubstrate. In the presence of naphthalene, [1-^{13}C] acenapthene was oxidized by strains BR and BC, with naphthalene no longer detectable after 24 hours. Only partial (30 to 40 percent) depletion of acenapthene occurred during the first 5 days of incubation. No further degradation of acenapthene was observed after longer incubation.

Mueller, J.G. et al., 1993
This laboratory study was designed to evaluate the ability of a sequential inoculation process using selected microorganisms to enhance the bioremediation technologies for the treatment of groundwater contaminated with creosote and PCP. The contaminated groundwater was obtained from a monitoring well at the American Creosote Works site in Pensacola, Florida. Both 1.2 L (bench scale) and 454 L (pilot scale) bioreactors were utilized for the analysis. The bench scale study showed that after 32 days of continuous-flow operation, the majority of the monitored creosote components were degraded. Overall, for groups 1, 2, and 3 PAHs, the biodegradation values were determined to be 98.0, 96.2, and 89.4 percent, respectively. The amount of group 2 and 3 PAHs found in the bioreactor residues were 3.5 and 9.2 percent, respectively. In the pilot scale study, the system was effective in treating the contaminated water in a two-step process. Because the pattern of degradation favored the low molecular weight components, an additional inoculum of microorganisms, selected for their ability to degrade these components, were added. As a result, more than 98 percent of all the monitored creosote components were removed. A mass balance distribution analysis showed that of the various routes of removal (adsorption, volatilization, and biodegradation), biodegradation was the primary mechanism for the removal of the creosote components.

National Oceanic and Atmospheric Administration, 1988

The National Oceanic and Atmospheric Administration (NOAA) conducted a detailed study on the analysis of PCBs, PAHs, and 12 trace metals present in surface water sediments of 200 estuaries. One hundred seventy-five estuaries were contaminated with PAHs. These estuaries have muddy rather than sandy sediments with the exception of two estuaries in Long Island, where the sediments were predominantly sandy. The results of the study indicated that PAH contamination was higher in the muddy sediment.

Bouwer, E.J., W. Zhang, L.P. Wilson, and N.D. Durant, 1996
A four-week aerobic study was conducted on the sediment and groundwater of an abandoned manufactured gas plant (MGP) site which was contaminated with creosote PAHs . The experiment was conducted to assess the ability of the bacteria in sediment to mineralize ^{14}C-labeled benzene, naphthalene, and phenanthrene under simulated field conditions. The results of this study showed that the bacteria present in the aquifer sediments were able to degrade low molecular weight PAHs. The study also determined that the higher the concentration of oxygen, the greater the biodegradation of these molecules. Mineralization for anthracene and phenanthrene ranged from 4 to 23 percent, ranged from 4 to 42 percent for benzene, and from 8 and 55 percent for naphthalene.

Chapman, P.J., M. Shelton, M. Grifoll, and S. Selifonov, 1995
In this study, bacterial cultures were obtained from washing creosote-contaminated soils at lumber treatment facilities. These cultures were to be utilized as an enrichment for biodegradation studies. Purified PAHs were used as the sole carbon sources (0.1%) and were inoculated with the bacterial cultures. Increases in turbidity and color changes were monitored to determine growth and biological activity, respectively. Low molecular weight PAHs were easily degraded by the bacterial enrichments. Approximately 72 percent of the measured PAH were degraded, which accounted for 52.5 percent of the weight of the initial PAHs. These aerobic degradation processes of creosote PAHs have shown that biodegradation in the presence of certain bacteria is accompanied with the formation of neutral and acidic oxidized products.

Schocken, M.J. and D.T. Gibson, 1984
The metabolism of acenaphthene and acenaphthylene by two strains of bacteria was examined in this study. *Biejerinckia species* and a mutant strain *Biejerinckia species* B8/36 bacteria were found to oxidize the two PAHs. The study was carried out with large-scale incubations at 30°C. Acenaphthene oxidized into 1-acenaphthenone, 1,2-acenaphthenediol, acenapthenequinone and 1-acenaphthenol; acenaphthylene oxidized into acenapthenequinone. The results indicated that even though these PAHs were both oxidized to acenapthenequinone, the pathways to form this product are quite different.

Godsy, E.M., D.F. Goerlitz, and D. Grbic-Galic, 1992
The US Geological Survey selected an abandoned wood preserving site once used for creosote and pentachlorphenol pressure treatments to analyze and identify the contaminants in groundwater from plant waste. This waste had been discharged into the unlined surface impoundments that were in direct hydraulic contact with groundwater. The groundwater was determined to be anaerobic and showed the presence of methane and hydrogen sulfide (indicating the presence of methanogenic and sulfuryl microbes). To conduct the study, holes were bored to a depth of 6.1 meters at various sites on a downward slope from the contamination source ponds. Table 4 lists the concentrations of PAHs at the various sites. The results indicated that lateral migration of most of the PAHs became undetectable 150 meters from the source of

contamination. However, the study authors did not indicate the age of the wood and how long the preserving plant had been abandoned before testing began.

Table 4. Amounts of PAHs in the Water Samples From 6.1 Meter Wells[a]

Compound[b]	Site 1[c] (uncontam.)	Site 3 (6.0 m)	Site 39 (53 m)	Site 40 (99 m)	Site 4 (122 m)	Site 37 (150 m)
Indene	ND	1.25	0.24	ND	ND	ND
Naphthalene	ND	9.38	3.39	2.89	0.93	1.54
1-Methylnaphthalene	ND	0.41	0.32	0.25	0.06	0.11
2-Methylnaphthalene	ND	0.99	0.32	0.54	0.10	0.10
Acenaphthene	ND	0.52	0.29	0.33	0.05	ND
Indole	ND	ND	ND	ND	ND	ND
Quinoline	ND	11.2	0.01	ND	ND	ND
Benzothiophene	ND	0.83	0.31	0.22	0.16	0.16
Dibenzofuran	ND	0.30	0.04	0.16	ND	ND

ND = Not detected
a - Water samples collected from wells on these sites.
b - Analyzed and quantitated through GC/MS methods.
c - Uncontaminated site

Goerlitz, D.F. et al., 1985
A similar study was conducted by the US Geological Survey at the same wood preserving plant that was used in the study of Godsy et al. (1992). Results showed that on Site 3 at a 30-meter well, naphthalene (15.60 ppm) was detected 6 meters from the source of contamination and the level of naphthalene progressively decreased with an increase in well depth. At 24 meters, only 0.60 ppm of naphthalene was detected. PAHs were not detected at a depth greater than 12 meters.

Genther, B.R., et al., 1997
A year long study on anaerobic soil biodegradation was conducted at the American Creosote Works Superfund site located in Pensacola, Florida using soil samples collected from the creek bed sediment. Samples were collected at a depth of 5 to 8 cm and 12 meters, which was below the main discharge pond. These soil samples were selected because they represented bacterial populations exposed to minimum and maximum concentrations of the PAHs *in situ*. The PAHs used were from an artificial mixture simulating the PAH components of creosote. Various batches of the artificial mixture were made and consisted of the following components: naphthalene (36 mg), 1-methylnaphthalene (10.8 mg), 2-methylnaphthalene (10.8 mg), 2,6-dimethylnaphthalene (10.8 mg), biphenyl (5.4 mg), acenaphthene (5.4 mg), fluorene (10.8 mg), phenanthrene (18.0 mg), anthracene (18.0 mg), 2-methylanthracene (9.0 mg), anthraquinone (3.6 mg), fluoranthene (9.0 mg), pyrene (3.6 mg), chrysene (3.6 mg), 2,3-benzo[b]fluorene (3.6 mg), and benzo[a]pyrene (3.6 mg). Various batches of this artificial PAH mixture were inoculated

with the contaminated soil samples under methanogenic, sulfidogenic, and nitrate-reducing conditions.

The majority of the PAHs did not degrade in the soil samples collected from the creek bed (5-8 cm depth). Loss of some bicyclic and tricyclics were observed; however, 4- and 5-membered PAHs did not degrade under methanogenic, sulfidogenic, or nitrate-reducing conditions. By 16 weeks, under methanogenic conditions, the maximum loss of naphthalene was 47 percent (similar to naphthalene loss under an abiotic control (40 percent)). Among the tricyclics, anthraquionone was the only substance where any loss under methanogenic conditions was reported. Anthraquinone loss reached 48 percent and about 65 percent by weeks 28 and 52, respectively. Two other tricyclics, anthracene and acenaphthene, degraded 22 percent under the same conditions. Under nitrate-reducing conditions, only degradation for 2-methylanthracene was observed and the concentration of this chemical was below the detection limit between weeks 8 to 28. Under sulfidogenic conditions, only anthraquinone degraded 22 percent by week 8. Under these conditions, no other PAHs showed any biodegradation up to week 52. For the 12-meter soil samples, no appreciable biodegradation processes were observed for most of the PAHs under all three anaerobic conditions.

Various Authors
Anaerobic soil metabolism studies for individual PAHs have been conducted over a period of time using the three anaerobic conditions. For nitrate reducing conditions, the studies conducted were Bouwer and McCarty, 1983; Al-Bashir, et al., 1990; Ehrlich et al., 1982a; Flyvberg et al. 1993; Hambrick et al., 1980; Kuhn et al., 1988; and Mihelcic and Luthy, 1988a, 1988b. The study was conducted for sulfidogenic conditions by Flyvjberg et al., 1993. The studies for methanogenic conditions include Ehrlich et al. 1982a; and Godsy et al., 1992. Other *in situ* studies for biodegradation of PAHs at creosote contaminated sites are: Ehrlich et al. 1982a, 1982b; Godsy et al. 1992; Goerlitz et al., 1985; and Mattraw and Frank, 1986.

Bauer, J.E. and D.G. Capone, 1985
Microorganisms present in the intertidal sediments were investigated for the degradation of anthracene and naphthalene. No mineralization was observed under anaerobic conditions. However, mineralization did show dependence on the amounts of the polyaromatics present, oxygen level, and pre-exposure time. Maximum mineralization of these two PAHs occurred after one or two weeks of pre-exposure. A similar study conducted by the same authors (in 1988) showed that the pre-exposure of anthracene and naphthalene under aerobic conditions (marine sediments collected from 0 to 1 cm depth) to benzene and other PAHs accelerated their mineralization.

Mueller, J.G. et al., 1991
Groundwater samples collected from a depth of 7 meters from the American Creosote Works superfund site in Pensacola, Florida were examined for biodegradation (aerobic aquatic metabolism) of PAHs from coal-tar creosote. Creosote-contaminated soil from the site was collected and used to prepare the microbial inoculum. Groundwater samples and the microbial inoculum were incubated for 14 days at 30°C. The groundwater samples contained the PAHs present in coal-tar derived creosote and in the phenolic components (which constituted 5% of the creosote sample). In addition, PCP was also present. Table 5 summarizes the results of the 14-day incubation experiment. The analysis showed that bicyclic PAHs and phenolics are metabolized readily by the microorganisms in the aquatic soil while the PAHs with a higher

number of fused rings biodegrade more slowly. By Day 8 most of the biodegradation was complete. A few nitrogenous heterocyclics like quinoline, isoquinoline, and acridine readily metabolized while quinaldine, carbazole, and pentachlorphenol showed resistance to the metabolic process under the conditions of the experiment.

Table 5. Aquatic Soil Metabolism (Biodegradation) Results for Creosote/PCP Contaminants In Groundwater Samples From Pensacola, Florida Superfund Site

Compound	Concentration (μg/mL) After Incubation						
	Initial Conc.	1 Day	3 Days	5 Days	8 Days	14 Days	Sterile Control
Naphthalene	28.7	17.2	0.1	U	0.1	U	25.6
2-Methylnaphthalene	4.7	3.0	U	U	0.1	U	4.5
1-Methylnaphthalene	9.5	5.7	2.1	1.5	U	U	8.2
Biphenyl	3.0	1.7	1.2	U	U	U	2.6
2,6-Dimethylnaphthalene	2.4	1.4	1.2	1.2	1.0	0.3	2.1
2,3-Dimethylnaphthalene	1.3	0.8	0.5	0.8	0.7	0.2	1.0
Acenaphthalene	0.6	0.3	0.4	0.6	0.6	0.2	0.4
Acenaphthene	13.6	9.0	8.3	9.6	9.7	1.8	11.9
Fluorene	11.6	7.8	8.0	5.2	1.8	0.1	9.9
Phenanthrene	32.8	23.5	23.1	15.4	0.3	U	27.7
Anthracene	4.7	3.2	3.0	2.7	2.2	0.5	3.9
2-Methylanthracene	5.2	3.7	3.7	4.0	4.2	1.5	4.4
Anthraquinone	3.3	2.7	1.9	U	U	U	2.9
Fluoranthene	16.2	11.5	11.5	13.3	13.5	7.6	14.4
Pyrene	10.4	7.8	7.3	8.2	8.3	4.7	9.8
Benzo[b]fluorene	2.5	1.7	1.7	1.8	2.0	1.2	2.0
Chrysene	2.7	1.8	1.8	2.0	2.1	1.2	2.4
Benzo[a]pyrene	2.1	0.5	U	U	U	0.9	2.0
Benz[a]anthracene	2.9	2.0	2.0	2.0	2.2	1.3	2.7
Benzo[b]fluoranthene/benzo[k]-fluoranthene	2.9	2.8	2.0	2.1	2.1	1.7	2.8
Indeno[1,2,3-c,d]pyrene	1.9	1.3	1.4	1.4	1.2	0.9	1.8
2,6-Xylenol	1.1	0.6	0.2	0.1	0.1	U	0.8
o-Cresol	4.2	2.7	0.3	0.2	0.2	U	4.9
2,5-Xylenol	0.1	U	U	U	U	U	0.1

Compound	Concentration (µg/mL) After Incubation						
	Initial Conc.	1 Day	3 Days	5 Days	8 Days	14 Days	Sterile Control
2,4-Xylenol	0.2	U	U	U	U	U	0.2
p-Cresol	2.0	0.1	U	U	U	U	2.3
m-Cresol	2.5	1.9	U	U	U	U	2.3
2,3-Xylenol	0.2	0.1	U	U	U	U	0.1
3,5-Xylenol	1.3	0.5	0.2	0.1	U	U	1.1
3,4-Xylenol/2,3,5-trimethylphenol	0.4	0.1	0.1	0.1	U	U	0.3
PCP	0.1	0.3	0.1	0.1	0.1	0.1	0.1
2-Picoline	0.3	0.2	0.2	0.2	0.1	U	U
3-Picoline/4-picoline	U	U	U	U	U	U	U
Lutidine	0.9	0.7	0.6	0.5	0.5	0.6	0.8
Thianaphthene	20.3	12.5	2.6	1.2	0.5	0.3	23.4
Quinoline	4.3	1.7	0.5	0.2	0.2	0.2	3.6
Isoquinoline	1.5	0.9	0.3	0.3	0.1	U	1.4
Quinaldine	3.4	3.2	2.8	2.4	0.6	0.3	4.9
Lepidine	0.7	0.6	0.4	0.3	0.3	0.2	0.7
Dibenzofuran	5.5	5.9	5.8	3.4	1.0	0.7	6.1
Dibenzothiophene	3.8	2.8	3.1	2.0	1.3	1.2	3.1
Acridine	22.5	18.2	14.1	1.9	2.0	2.0	26.2
Carbazole	2.9	2.1	1.2	0.9	0.8	1.0	3.0

Note: Reported data are averages of duplicate samples.
U - Below the detection limit.

Hurst, C.J. et al., 1996
An aerobic and anaerobic soil metabolism study was conducted at the Champion International superfund site in Libby, Montana. The contaminated soil samples were spiked for analysis. Biodegradation for ^{14}C-pyrene and seventeen PAHs (including pyrene) was followed by 0, 2, and 21 percent oxygen as soil gas. The PAHs chosen for the study were: naphthalene, acenaphthylene, acenaphthene, fluorene, anthracene, phenanthrene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a] pyrene, dibenzo[a,h] anthracene, benzo[g,h,i]perylene, and indeno(1,2,3-cd). After 70 days, 45 to 55 percent of the ^{14}C-pyrene was mineralized in 2 percent and 21 percent oxygen. At 0 percent

oxygen, no statistically significant mineralization was observed. For eight of the non-radiolabeled PAHs, biodegradation in an oxygen atmosphere ranged from 6.2 percent (naphthalene) to 57 percent (pyrene). The remaining PAHs were below the limit of detection.

Leaching and Adsorption/Desorption

Villholth, K.G., 1999
The objective of this study was to determine the amount of colloids in the groundwater of two creosote-contaminated aquifers in Denmark and to determine the *in situ* distribution of the PAHs between the dissolved and colloidal phases in the water. The colloids identified at the sites were clay minerals, iron-oxides, iron-sulfides, and quartz particles containing significant amounts of organic carbon. The results of a two-step fractionation procedure, showed that the PAHs partitioned to the course (>100 nm) colloid fraction (log K_{oc}) and was linearly correlated with the PAH octanol-water partition coefficient (log K_{ow}), indicating the partitioning was hydrophobic. This suggested a potential for colloid-facilitated transport of PAHs. The lack of PAH partitioning to colloids <100 nm, indicates a weaker binding to the smaller, more hydrophilic colloids in the groundwater.

Rutherford, P.M., et al., 1997
This study was designed to determine if a 10-week slurry phase bioremediation treatment altered the desorptive properties of two-creosote-contaminated soils. Soil samples were collected from an inactive wood preserving facility in Edmonton, Alberta (EDM site) and Prince Albert, Saskatchewan (PAA site). For bioremediation, soil samples were combined with aqueous nutrient media under aerobic conditions in a bioreactor. Desorption of ^{14}C-naphthalene from the contaminated soils was measured before and after bioremediation in a sequential batch experiment. The contributions of the contaminant organic phase and the soil organic matter to desorption were determined by experiments on soils with and without nonaqueous phase liquid contaminants. Results showed that total extractable organics were 43 to 31 percent lower in the bioremediated soils for both sites. This reduction in total organic carbon content lowered the sorption capacity of the soils. The desorptive partition coefficient (K_d) for the nonbioremediated EDM soil was significantly ($p = 0.022$) greater than the nonbioremediated PAA soil. However, after bioremediation, no significant difference was found between the two soils ($p = 0.11$). Although the K_d decreased due to bioremediation, the carbon-based partition coefficients on the nonaqueous phase liquid did not change significantly once the changes in the overall composition of the soil had been accounted for.

Priddle, M.W. and K.T.B. MacQuarrie, 1994
A laboratory study was conducted to evaluate the dissolution of industrial creosote in water using a generator column to determine the impacts on groundwater quality and the kinetics of the dissolution process. The laboratory results were also compared to an equilibrium model and a quasi-kinetic model. For the laboratory study, the generator column was packed with a 10 percent creosote effluent obtained from Carbochem Inc. (Hamilton, Ontario, Canada). The study focused on 10 specific PAHs including naphthalene, phenanthrene and benzo[a]pyrene. A mass-transfer rate test was conducted to evaluate the rate the creosote components reached equilibrium concentrations. Additionally, two long-term dissolution tests were conducted by passing water through the column for a designated period of time. The mean contact time in the column was

0.56 and 1.02 hours during the two testing periods. The results of the mass transfer test indicated that the creosote components reached equilibrium with the aqueous phase in about 60 hours. The two long-term tests found that the concentrations of the targeted PAHs detected in the effluent all decreased steadily throughout the monitoring period. Initial concentrations of the components were approximately 40 percent of the calculated effective solubilities. The higher molecular weight compounds were not detected which was expected due to the low effective solubilities (<0.002 mg/L^{-1}). Overall, the ratios of these concentrations were in proportion to their effective solubilities which were calculated using Raoult's law and the creosote composition data.

Padma, T.V. et al., 1999
A study was conducted to monitor the effects that various processes (tides, storms, bioturbation, shipping, and dredging) may have in the dissolution and resuspension of sediment-associated PAHs. These environmental processes were mimicked by creating a water-soluble fraction from the creosote-contaminated sediment and artificial estuarine water. Creosote-contaminated sediment samples were collected near Atlantic Wood Industries on the Southern Branch of the Elizabeth River in Virginia. The results showed that the creosote-contaminated sediment source contained more intermediate weight (three aromatic rings) and high molecular weight (more than three aromatic rings) PAHs, in contrast to the water-soluble fraction, which contained high levels of low molecular weight (less than three aromatic rings) and heterocyclic compounds. These differences were due to fractionation and degradation of creosote in the water soluble fraction.

EPRI, 1992
EPA's toxicity characteristic (TC) rule identifies three phenolic isomers (o-, m-, and p-cresol) as regulated substances. The Electric Power Research Institute (EPRI, 1992) used the TCLP, a leaching method, to determine the concentrations of the three phenolic isomers that leached from treated wood poles and crossarms when nearing disposal. According to the EPA regulation, materials that leach more than 200 mg/L are classified as hazardous and can not be disposed of as solid waste into landfills. Fifty four samples from seventeen poles and six crossarms, (ages 7 to 53 years) were chosen from the Northeast, Mid-Atlantic, Southeast, Midwest, North Central and Western regions of the United States. Southern Pine, Douglas Fir, Western Red Cedar, and Cedar woods were chosen for the study. Total cresol concentration (all three isomers) present in the leachates ranged from below the detection limit (0.01 mg/mL) to 14.95 mg/mL, and the mean concentration was 1.63 mg/mL. This was below the Agency's toxic characteristic regulatory level of 200 mg/mL.

Gile, J.D. et al., 1982
Seven Ponderosa pine blocks (3.3 x 2.6 x14 cm) pressure-treated with radiolabeled phenanthrene, acenaphthene, and bis(tri-n-butyl oxide) were tested in a terrestrial microcosm chamber (TMC-II). This microcosm contained Williamette Valley topsoil, ryegrass, invertebrates, and a gravid gray-tailed vole. The impregnation mixture contained dieldrin as a reference compound. The study was conducted for 2.5 months at which time it was found that 95 percent of the pesticides remained in the wood and most of the materials that leached remained in the upper soil layer immediately surrounding the pine blocks. In the plants of the microcosm, 0.7 ppm of dieldrin was detected while phenanthrene was detected at a 8.8 ppm. Bioaccumulation in the invertebrates was variable and the concentration of phenanthrene in the vole body was 7.2 ppm, while acenaphthene was detected at 37.0 ppm.

Wendt, P.H., 1996
In a six-week two-phase field study, private, residential docks located on ten tidal creeks at South Carolina's Charleston Harbor Estuary treated recently with CCA, but originally treated with creosote were studied. Samples were collected from sediments and oysters (*Crassostrea virginica*) <1 meter, and >10 meters from the docks. Reference samples were also collected. Mean concentrations of the 12 PAHs monitored from the sediment were 978.3 µg/kg (dry wt.), 690.0 µg/kg (dry wt.), and 1183.8 µg/kg (dry wt.) for the <1 m, >10 m, and the reference samples, respectively. Mean PAH concentrations from the oysters were 3547.3 µg/kg (dry wt.), 2057.6 µg/kg (dry wt.), and 2173.1 µg/kg (dry wt.) for the <1 m, >10 m, and reference samples, respectively. The study author reported that these concentrations were generally within the range of values found at nearby marinas. Most concentrations of PAHs in whole sediments were generally below Long et al.'s (1995) "ER-L" (Effects Range-Low) levels, suggesting that these values reported were insufficient to cause any adverse biological/toxic effects.

Bestari K.T. Jim et al., 1998a
In a field study, the fate of creosote components in aqueous medium, sediment, and adsorption processes (using PVC strips) were studied using fourteen microcosms (12,000 L volume) filled with sediment consisting of 53 percent sand/gravel, 25 percent silt, and 22 percent clay. The microcosms were treated with liquid creosote in concentrations ranging from 0.06 ppm to 109 ppm. The concentrations for 15 PAHs (recognized by the Agency as the priority pollutants) showed a rapid decline in the water. Two days after the application, total PAHs measured in the water were 7.3 :g/L (0.06 ppm application) and 5,803.2 :g/L (109 ppm application). By Day 84, total PAHs remaining in the water were 0.80 :g/L and 13.9 :g/L from the 0.06 ppm and 109 ppm applications, respectively. In sediments, total PAHs ranged from 0.91 :g/g to 63.9 :g/g at Day 28, then declining thereafter. A similar trend of declining concentrations was shown by the PAHs on the PVC strips. A mass balance calculation reported a loss of 88.3 percent of PAHs from the microcosms after one month. Based on the total PAH concentrations, the half-life of most PAHs in water was reported to be approximately one week.

Bestari, K.T. Jim et al., 1998b
Marine-grade Douglas fir pilings (15 to 20 cm diameter and 1.2 m length) were pressure-treated with creosote using the same concentrations (0.06 ppm to 109 ppm) as in Bestari K.T. Jim et al. (1998a) in a similar simulated microcosm. This microcosm contained sediment, rooted and floating macrophytes, and fish and invertebrate communities which consisted of phytoplankton, zooplankton, and benthos. The total organic content was 5.1 percent in water that had been circulated in the microcosm through a holding tank for four weeks to maximize the chemical and biological compositions. The pressure-treated pilings were suspended vertically in each microcosm in such a way that they were just above the water surface and not touching the sediment.

The experiment was conducted for one year beginning in 1994. Water samples were collected prior to the creosote treatment (Day 1), Day 2, and Day 5 after the treatment, then weekly up to four weeks and biweekly thereafter for twelve weeks. Quantitative analyses were performed on these solutions. The analysis showed that the concentration of PAHs increased rapidly through Day 7 after the treatment (7.3 :g/L to 97.2 :g/L) and then declined to concentrations close to the background (0.80 to 6.7 :g/L) by the end of the study (Day 84). Total PAHs from the leachates did bind to the PVC liner, concentrations on Day 31 ranged from 0.3 to 2.4 : g/cm^2, and ranged from 0.2 to 2.2 : g/cm^2 58 days after the treatment. The rate of loss of creosote from the pilings was 50 : g/cm^2/day. The study suggested that the rapid loss of creosote was primarily due to degradation processes like photolysis and microbial decomposition, and partial adsorption to PVC liners. The PAHs identified in this and previous experiments were: naphthalene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benz[a]anthracene, chrysene, benzo[b]pyrene, benzo[k]fluoranthene, benzo[a]pyrene, benzo[ghi]perylene, indeno[1,2,3-cd]pyrene and dibenz[a,h]anthracene. These are fifteen of the sixteen PAHs that EPA has recognized as EPA Priority Pollutants.

Wan, M.T., 1994

In this study, runoff water from pressure-treated utility and telephone wood poles was collected from fourteen utility and six railway ditches. The utility and telephone poles were initially pressure-treated with pentachlorophenol and later *in situ* treated with a mixture of creosote and chlorophenols. The treatment was typically at the base up to 0.5 m above and below soil level, and these applications were either used as wrappings or painted with a creosote/chlorophenol mixture. The runoff water was analyzed for the presence of 15 PAHs. These right of way (ROW) ditches which were sampled flow into salmon streams in the Lower Mainland and Vancouver Island of British Columbia, Canada. Ditches of parklands, farmlands, and railway ROWs were also sampled to establish background and reference PAH concentrations. PAHs were not detected in the parkland ditches; however, they were found in farmlands and in utility and railway ROW ditches. The data from this study showed that the maximum concentrations of the PAHs are in the wood and the PAHs that leach out of the wood were mostly present around the base of the poles. Tables 6 through 8 summarize the presence/absence of the PAHs in various environmental compartments.

Table 6. PAHs in Treated Wood

Compound	Poles (mg/kg)	Railway Ties (mg/kg)
Acenaphthene	3,566	1,238
Acenaphthylene	244	77
Anthracene	17,686	5,100
Benz[a]anthracene	1,938	465
Benz[b+k]fluoranthene	811	254
Benz[a]pyrene	1,116	461
Benz[ghi]perylene	69	37
Chrysene	1,093	383
Dibenz[a,h]anthracene	7	16
Fluoranthene	7,832	1,880
Fluorene	5,313	1,141

Indeno[1,2,3-cd]pyrene	36	40
Naphthalene	1,514	342
Phenanthrene	15,378	3,173
Pyrene	5,289	1,487
Total PAHs	61,182	16,094

Note: These PAHs were extracted from a sample of wood chips/scraps which were collected and made into a composite mixture of 25 poles or 25 railway ties.

Table 7. PAHs in Ditch Water of Parklands, Farmlands and Railway Rights-of-Way in the Lower Mainland of British Columbia

Compound	Parkland Ditches (:g/L)	Farm Ditches (:g/L)	Railway Ditches (w/poles, :g/L)	Railway Ditches (w/o poles, :g/L)
Acenaphthene	ND	1.04	206	0.57
Acenaphthylene	ND	1.72	5.5	1
Anthracene	ND	0.16	81	01.3
Benz[a]anthracene	ND	0.10	195	0.12
Benz[b+k]fluoranth-ene	ND	0.19	144	0.19
Bnez[a]pyrene	ND	0.10	43	0.10
Benz[g,h,i]perylene	ND	0.19	12.2	0.14
Chrysene	ND	0.10	228	0.17
Dibenz[a,h]anthrace-ne	ND	0.26	4.1	0.10
Fluoranthene	ND	0.30	2035	0.26
Fluorene	ND	0.30	116	0.22
Indeno[1,2,3-cd]pyrene	ND	0.16	17.6	0.15
Naphthalene	ND	0.35	8.5	0.19
Phenanthrene	ND	0.40	1027	0.44
Pyrene	ND	0.19	1233	0.19
Total PAHs	ND	5.56	5,356.3	3.97

Note: In all cases the number of sample sites was never more than two.
ND = Not detected.

Table 8. PAH Concentrations in Utility ROW Sediments in the Lower Mainland of British Columbia

Compound	Concentration (µg/L)			
	Base of Pole	Ditches 4 m Upstream of Pole	Ditches Adjacent to Pole (0.1-0.3 m)	Ditches 4 m Downstream of Pole
Acenaphthene	221	0.11	1.03	ND
Acenaphthylene	36	ND	1.37	ND
Anthracene	706	0.16	2.12	0.09
Benz[a]anthracene	93	0.05	0.46	0.25
Benz[b+k] fluoranthene	71	ND	0.65	0.73
Benz[a]pyrene	67	0.11	0.41	0.64
Benz[g,h,i]perylene	73	ND	0.43	0.20
Chrysene	92	0.11	0.41	0.17
Dibenz[a,h]anthracene	91	ND	0.20	0.27
Fluoranthene	211	0.19	1.24	0.24
Fluorene	469	0.10	1.58	ND
Indeno[1,2,3-cd]pyrene	75	0.25	0.71	0.34
Naphthalene	34	0.12	0.08	ND
Phenanthrene	666	0.07	3.29	0.12
Pyrene	171	0.06	1	0.22
Total PAHs	3076	1.33	15	3.27

Note: Sampling size varied from 5 to a maximum of 8 samples.
ND = Not detected

Middaugh, D.P. et al., 1991
An Agency sponsored study was conducted on the leaching of creosote components from an abandoned American Creosote Works Site to a freshwater stream that flows into the Florida Pensacola Bay. This site also utilized pentachlorophenol (PCP) and chromated copper arsenate (CCA). Adjacent to the site, a well was dug to a depth of 21 meters and the ground water was analyzed. The PAH concentrations in groundwater were reported as follows: phenanthrene (32.8 mg/kg), naphthalene (28.8mg/kg), fluoranthene (16.1 mg/kg), acenaphthene (13.6 mg/kg), fluorene(11.6 mg/kg), pyrene (10.4 mg/kg), 1-methyl naphthalene(9.5 mg/kg), and 2-

methylanthracene (5.2 mg/kg). The concentrations of other PAHs detected were either less than 5 percent of the ones noted above or were very low.

Bioaccumulation

Stegman and Teal, 1973 and Varsani, et al., 1978
Studies on bioaccumulation showed that in an aquatic habitat, organisms such as fish, shellfish, and crustaceans readily accumulate PAHs from the environment and store them at a high level in their tissues.

Southworth, G.R., etal., 1978
A study was designed to investigate the bioaccumulation of seven selected PAHs in *Daphnia pulex*. The PAHs selected were: naphthalene, anthracene, phenanthrene, pyrene, 9-methylanthracene, benz[a]anthracene, and perylene. Bioaccumulation corresponded to the octanol/water partition coefficients (K_{OW}). The benz[a]anthracene bioaccumulation factor was 10,000 fold higher and about 100 fold higher for naphthalene.

A bioaccumulation study on clams (*Rangia cuneate*) was also conducted after a creosote spill into Bayou Bonfuca at the American Creosote Works Plant site at Slidell, Louisiana. Results showed that the levels of PAHs increased gradually for two weeks and increased dramatically by week four at the monitoring station closest to the spill site. PAH results are shown in Table 9.

Another station (control station) in the same study showed evidence of depuration after two weeks and equilibration after four weeks. The concentration of PAHs in water was very low at the site close to the spill (13 ppb in two weeks and 26 ppb after four weeks). Among the PAHs, benzopyrenes were detected at a very high level of 600 ppb at the station closest to the spill site. Other studies (Politzer, 1985; Neff, 1976) also showed that depuration of PAHs in bivalves vary from a couple of weeks to a few months. These studies support the possibility of bioavailability of PAHs and contamination of the food chain.

Table 9. Amounts of PAHs Detected in Clams at the Closest Station to the Spill Site

PAH Component	Week 0 - Pre-exposure (ppb)	Week 2 (ppb)	Week 4 (ppb)
Naphthalene	43	60	120
Biphenyl/Acenapthylene	17	13	42
Fluorene	7	5	11
Phenanthrene/ Anthracene	34	10	28
Fluoranthene/Pyrene	120	88	130
Benz[a]anthracene/Chrysene	41	81	190
Benzopyrenes	87	132	600

Spacie, A., et al., 1983
Bluegill sunfish were used to investigate the uptake (bioaccumulation), biotransformation, and depuration rates of anthracene and benzo[a]pyrene. The uptake half-life for anthracene was 0.019 hours and did not appear to be affected by the exposure concentration and humics. The uptake half-life of benzo[a]pyrene was also similar, but was affected by the presence of humics. Biotransformation for anthracene was determined to be 0.22 nmol/hr while for benzo[a]pyrene varied from 0.044 nmol/hr to 0.088 nmol/hr, between 1 and 2 hours of exposure. The depuration half-life for anthracene was 17 hours and 67 hours for benzo[a]pyrene. Due to the biotransformation, the bioconcentrations for both anthracene and benzo[a]pyrene were lower than predicted from the K_{ow}.

Tay, K.L., et al., 1992
A detailed bioassessment study was conducted in Canada on the Halifax Harbor Sediment. Bioaccumulation of PAHs and other organic and inorganic contaminants were investigated on the bivalve mollusk (*Macoma balthica*). The exposure of the species to the contaminated sediment was 30 days. The study authors suggested that the duration of 30 days may have been too short for uptake of the PAHs into the *Macoma* system. Organic content in the Tuft's Cove sediment was high and that may have prevented the bioaccumulation process. Table 10 summarizes the results.

Table 10. Bioaccumulation of PAHs in *Macoma balthica*

Station	PAHs in Tissues (mg/kg wet wt.)
Walton (Control sediment)	0.11
Drakes Gut 1 (Reference sediment)	0.13
Tuft’s Cove (Contaminated sediment)	ND
Original tissues (not exposed to test sediment)	0.01

J.F. Elder and P. Dresler, 1988
A study on bioaccumulation was conducted on mollusk species at the Pensacola Bay, 500 m from the creosote wood-preserving facility in Pensacola, Florida. Only four PAH compounds (phenanthrene, fluoranthene, pyrene, and naphthalene) were found in the mollusk species studied. Depuration rates were not reported. The same study also estimated the PAH presence in water and the surface layer of estuarine sediments. The study, conducted on seven sites, found that the sediments of the drainage streams were heavily contaminated with PAHs. The analysis of sediments at the sites showed very little contamination with PAHs except at one site (Site 4). This site was closest to the wood-preserving facility. The same study also reported that the bioaccumulation of fluoranthene, pyrene and phenanthrene in both species of mollusks was ten times higher at the test site than at the control site. Table 11 summarizes the results.

Most of the PAHs were insoluble in water and the solubilities were inversely related to the molecular weights of the polyaromatics. Bioaccumulation of a pesticide depended on many external factors such as resistance to biodegradation, chemical degradation, photolysis, tendency for migration, and bioavailability. The data on bioaccumulation of PAHs are not extensive; however, a few generalities emerged from the existing data. First, molecules with lower molar masses had a tendency to bioaccumulate more than high molar mass substances. Second, bioaccumulation was also dependent on the concentrations of a substance. For example, between naphthalene (two fused-ring compound) and phenanthrene (three fused-ring compound), it is the later which was found to bioaccumulate more than naphthalene because in the original mixture of PAHs, phenanthrene was 20 percent of the mixture while naphthalene was 3 percent.

Table 11. Concentrations of PAHs in the Sediment and Discharge Stream at Pensacola Bay

PAH	Stream Site 3 (g/kg)	Stream Site 2 (g/kg)	Pensacola Bay Site 4 (g/kg)	Pensacola Bay/ Other Sites (g/kg)
Naphthalene	300	200	ND	ND
Phenanthrene	ND	12000	ND	ND
Fluoranthene	62000	17000	190	ND
Pyrene	32000	11000	160	ND
Benzoanthracene	15000	5000	75	ND
Chrysene	10000	7000	100	ND
Acenaphthene	19000	5000	ND	ND
Fluorene	32000	3000	ND	ND
Anthracene	140000	3000	ND	ND

ND - Not detected. The limit of detection was 40 g/kg.

Bruner, K.A. et al., 1994
To measure PAH bioconcentration factors, a study was conducted on pre-spawning (high lipid) and post-spawning (low lipid) mussel populations of zebra mussel in the Great Lakes. Pre-spawning mussels had greater bioconcentration factors and a faster rate of accumulation for benzo[a]pyrene than post-spawning mussels. Lipid content, however, did not influence bioconcentration factors or the rates of accumulation. Rates of depuration were not influenced by either factor (high or low lipid contents).

Hellou, J.G., et al., 1990
A study was conducted on the bioaccumulation of PAHs in marine mammals. Four species of seals and six species of whales from the waters around Newfoundland and Labrador were utilized as test subjects. Accumulated values, when expressed in terms of chrysene-equivalents were 0.10 to 1.21 ppm in the muscles of these mammals on a dry weight basis.

Howard, P.H. et al., 1991
It was suggested, based on theoretical calculations and modeling, that the half-lives of the PAHs can be estimated in air, water, soil and sediments. This data are presented in Table 12. From the table, one can arbitrarily divide the PAHs into three groups: PAHs with two aromatic fused rings, with three aromatic fused rings, and 4-5 aromatic fused rings. The study authors came to the following conclusions: 1) The half-lives of the PAHs in these environmental compartments increased as the complexity of the molecules increased. Generally, half-lives of 2 aromatic fused rings < 3 aromatic fused rings < 4/5 aromatic fused rings; 2) K_{ow} values also show a similar trend: three sets of K_{ow}s were observed. The K_{ow} values ranged from 3 to 4, from 4 to 5, and 6 and above. As noted for the half-lives, the Log K_{ow} increased as the complexity of the molecule

increased; 3) In general, the half-lives in air and water environmental compartments were lower than in soils/sediments, since the adsorption constant in these two compartments were larger than in air and water media. Those molecules with a longer half-life also exhibited persistence in that environmental compartment. PAHs were more persistence in soils/sediments than in other environmental compartments; and 4) The 4/5 aromatic-fused ring molecules were persistent and because they also had high K_{ow}s, they were also bioaccumulative in the organisms present in the soil/sediments.

Table 12. Estimated/Modeled PAH Half-lives in Air, Water, Soil and Sediment

Compound	Air/ Class	Half-live	Water/ Class	Half-life	Soil/ Class	Half-life	Sediment/ Class	Half-life	K_{ow}
Indan	2	1 day	4	1 wk.	6	2 mos.	7	8 mos.	3.33
Naphthalene	2	1 day	4	1 wk.	6	2 mos.	7	8 mos.	3.37
1-methyl naphthalene	2	1 day	4	1 wk.	6	2 mos.	7	8 mos.	3.87
Acenaphthalene	3	2 days	5	3 wks.	7	8 mos.	8	2 yrs.	3.92
Fluorene	3	2 days	5	3 wks.	7	8 mos.	8	2 yrs.	4.12
Phenathrene	3	2 days	5	3 wks.	7	8 mos.	8	2 yrs.	4.57
Anthracene	3	2 days	5	3 wks.	7	8 mos.	8	2 yrs.	4.54
Pyrene	4	1 wk.	6	2 mos.	8	2 yrs.	9	~ 6 yrs.	5.18
Fluoranthene	4	1 wk.	6	2 mos.	8	2 yrs.	9	~ 6 yrs.	5.22
Chrysene	4	1 wk.	6	2 mos.	8	2 yrs.	9	~ 6 yrs.	1.65
Benz[a]anthracene	4	1 wk.	6	2 mos.	8	2 yrs.	9	~ 6 yrs.	5.91
Benzo[k]flouranthene	4	1 wk.	6	2 mos.	8	2 yrs.	9	~ 6 yrs.	6.00
Benzo[a]pyrene	4	1 wk.	6	2 mos.	8	2 yrs.	9	~ 6 yrs.	6.04

Migration of PAHs From Poles to Soils

Mississippi State University, 1981

Fifty-six soil samples were collected radially around fourteen creosote-treated poles and analyzed for migration of creosote components into the surrounding soils. The concentrations of naphthalene, 2-methylnaphthalene, 1-methylnapthalene, and biphenyl varied from 25 to 50 Og/g. Acenaphthylene, acenaphthene, dibenzofuran, fluorene, phenanthrene, and anthracene were present at lower concentrations than in the original mixture of creosote. The author suggested that vaporization, water solubility, and biological degradation might be contributing factors for reduction in the concentrations of these polycyclic components of creosote. Fluoranthene, pyrene, carbazole, 1,2-benzanthracene, and chrysene were persistent in the soil around the poles and the concentration of all the components decreased as the distance from the poles increased.

McGroddy, S.E. and J.W. Farrington, no date available
In a study on sediment porewater partitioning of PAHs in Boston Harbor, it was shown that the PAH concentrations measured in sediments and porewaters from three cores were notably lower than the amounts of the PAHs predicted by two- and three-phase equilibrium partitioning models. The study author suggested lower amounts might have been available for partitioning in sediment porewaters.

LITERATURE REFERENCES

Al-Bashir, B. et al. 1990. Appl. Microbiol. Biotech. 34:414-419.

Bauer, J.E. et al. 1985. Appl. Environ. Microbiol. 81-90.

Bauer, J.E. et al. 1988. Appl. Environ. Microbiol. 1649-1655.

Behymer, T.D. and R.A. Hites. 1985. Environ. Sci. Technol. 19(10):1004-1006.

Bestari, K.T. Jim et al. 1998a. Distribution and composition of polycyclic aromatic hydrocarbons within experimental microcosms treated with liquid creosote. Environ. Toxicol. Chem. 17(12):2359-2368.

Bestari, K.T. Jim et al. 1998b. Distribution and composition of polycyclic aromatic hydrocarbons within experimental microcosms treated with creosote-impregnated douglas fir pilings. Environ. Toxicol. Chem. 17(12):2369-2377.

Bieri, R.H. et al. 1986. Polycyclic aromatic hydrocarbons in surface sediments from the Elizabeth River subestuary. Int. J. Environ. Anal. Chem. 26:97-113.

Bouwer, E.J. and P.L. McCarty. 1983. Appl. Environ. Microbiol. 45:1295-1299.

Bouwer, E.J. et al. 1996. Annals of New York Academy of Sciences. pp. 103-115.

Bruner, K.A. et al. 1994. J. Great Lakes Research. 20:725-734.

Brenner R.C. et al. 2002. Characterization and FATE of PAH-contaminated sediments at the Wyckoff/Eagle Harbor Superfund Site. Environ Sci Technol. 2002 Jun 15;36(12):2605-13.

Callahan, M.A. et al. 1979. Water Related Environmental Fate of 129 Priority Pollutants, EPA-440-4-79-029a,b.

Chapman, P.J. et al. 1995. Fossil Fuel Biodegradation: Laboratory Studies. Environ. Health Perspec. 103, Supplemental 5:80-83.

Elder, J.F. and P. Dresler. 1988. Accumulation and bioconcentration of polycyclic aromatic hydrocarbons in a nearshore estuarine environment near a Pensacola (Florida) creosote contamination site. Environmental Pollution. 49:117-132.

Electric Power Research Institute (EPRI). 1992. Document EPRI TR-01000870.

Ehrlich, G.G. et al. 1982. Groundwater. 20(4):703-710.

Flyvberg, J.E. et al. 1993. J. Contamin. Hydrology. 12:133-150.

Fowler, M.G. et al. 1993. Preliminary results from a field experiment investigating the fate of some creosote components in a natural aquifer. Org. Geochem. 22:641-649.

Fukuda, K. et al. 1988. Chemosphere. 17(4):651-659.

Gevao, B. and K.C. Jones. 1998. Kinetics and potential significance of polycyclic aromatic hydrocarbon desorption from creosote-treated wood. Environ. Sci. Tech. 32:640-646.

Gile, J.D. et al. 1982. J. Agric. Food Chem. 30:295-301.

Godsy, E.M. et al. 1992. Ground Water. 30(2):232-242.

Goerlitz, D.F. et al. 1985. Environ. Sci. Tech. 19(10):955-961.

Grbic-Galic, D. et al. 1991. Anaerobic degradation of aromatic hydrocarbons and aerobic degradation of trichloroethylene by subsurface microorganisms. In: Organic Substances and Sediments in Water, R.A. Baker, Ed., Lewis Publishers, Michigan:239-266.

Harris, J.C. 1982. Handbook of Chemical Property Estimation Methods. W.J. Lyman, W.F. Reehl, D.H. Rosenblatt, Eds. McGraw-Hill Book Company, New York: Chapters 7-8.

Hellou, J.G. et al. 1990. Polycyclic aromatic hydrocarbons in muscle tissue of marine mammals from the Norhtwest Atlantic. Marine Pollution Bulletin. 21(10):469-473.

Howard, P.H., R.S. Boethling, W.F. Jarvis, W.M. Meylan, E.M. Michalenco, Eds. 1991. Handbook of Environmental Degradation Rates, Lewis Publishers, Inc. Chelsea, Michigan, USA.

Hurst, C.J. et al. 1996. Polycyclic aromatic hydrocarbon biodegradation as a function of oxygen tension in contaminated soil. J. Haz. Materials. 51:193-208.

Karthikeyan, R. and A. Bhandari. 2001. Anaerobic biotransformation of aromatic and polycyclic aromatic hydrocarbons in soil microcosms: a review. Journal of Hazardous Substance Research. 3:3-19.

Kirso, U. et al. 1991. Photochemical oxidation of PAH and heteroaromatic analogues in different model conditions. Polycyclic Aromatic Hydrocarbons: Proceedings of the Thirteenth International Symposium on Polynuclear Aromatic Hydrocarbons.

Kuhn, E.P. et al. 1988. Appl. Environ. Microbiol. 54:490:496.

Lindhardt, B. and T.H. Christensen. 1996. Volatilisation of aromatic hydrocarbons from soil: part II, fluxes from coal tar contaminated soils residing below the soil surface. Water Air Soil Pollution. 92:375-389.

Long, E.R. et al. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environ. Management. 19:81-97.

Mabey, W. et al. 1982. Aquatic Fate Process for Organic Priority Pollutants. EPA Report No. 440/4-81-14.

Mattraw, H.C. and B.J. Frank. 1986. Movement and fate of creosote waste in ground water, Pensacola, Florida; U.S. Geological Survey toxic waste -- ground-water contamination program. U.S. Geological Survey Water Supply Paper No.: 2285.

McGroddy, S.E. and J.W. Farrington. Sediment porewater partitioning of polycyclic aromatic hydrocarbons in three cores from Boston Harbor, Massachusetts. Environ. Sci. Technol. 29(6):1542-1550.

Merril, E.G. and T.L. Wade. 1985. Carbonized coal products as a source of aromatic hydrocarbons to sediments from a highly industrialized estuary. Environ. Sci. Technol. 19:597-603.

Middaugh, D.P. et al. 1991. Archives of Environ. Contamin. pp. 244-254.

Mihelcic, J.R. et al. 1988. Appl. Environ. Microbiol. 54:1182-1187.

Mill, T. et al. 1981. Chemosphere. 10:1283-1293.

Miller, M.M. et al. 1985. Environ. Sci. Technol. 19:522-529.

Mississippi State University. 1981. Cooperative Agreement Number 12-156.

Mueller, J.G. et al. 1991. Biodegradation of creosote in contaminated groundwater: chemical and biological assessment. Applied and Environmental Microbiology. 57(5):1277-1285.

Mueller, J.G. et al. 1993. Strategy using bioreactors and specially selected microorganisms for bioremediation of groundwater contaminated with creosote and pentachlorophenol. Environ. Sci. Technol. 27(4):691-698.

National Oceanic and Atmospheric Administration (NOAA). 1988. A Selected Summary of Data of Chemical Contaminants in Sediments Collected During 1984-1987, NTIS.

Neff, J.M. et al. 1976. Marine Biology. Volume 38:279-289.

Padma, T.V. et al. 1999. Toxicity of creosote water-soluble fractions generated from contaminated sediments to the bay mysid. Ecotoxicology and Environmental Safety. 42:171-176.

Politzer, I.R. et al. 1985. Impact on Human Health of Petroleum in the Marine Environment, American Petroleum Institute (API), Washington, DC.

Priddle, M.W. and K.T.B. MacQuarrie. 1994. Dissolution of creosote in groundwater: an experimental and modeling investigation. Journal of Contaminant Hydrology. 15:27-56.

Rutherford, P.M., M.R. Gray, and M.J. Dudas. 1997. Desorption of [^{14}C]naphthalene from bioremediated soils contaminated with creosote compounds. 31:2515-2519.

Selifonov, S.A. et al. 1998. Use of 13C nuclear magnetic resonance to assess fossil fuel biodegradation: Fate of (1-13C)acenaphthene in creosote polycyclic aromatic compound mixtures degraded by bacteria. Applied and Environmental Microbiology; 64 (4). 1998. 1447-1453.

Sharak Genther, B.R. et al. 1977. Arch. Environ. Contamin. Toxicol. 32:99-105.

Shocken, M.J. et al. 1984. Bacterial oxidation of the polycyclic aromatic hydrocarbons acenaphthene and acenaphthylene. Appl. Environ. Microbiol. 48(1):10-16.

Smith, J.H. et al. 1978. Environmental Pathways of Selected Chemicals in Fresh Water Systems: Part II. Laboratory Studies: 304. EPA-600/7-78-074, USEPA, Athens, Georgia.

Southworth, R.G. 1977. Aquatic Toxicology, ASTM ATP 667. L.L. Marking, R.A. Kimerle, Eds., American Society for Testing and Materials: 359-380, Philadelphia.

Southworth, G.R., J.J. Beauchamp, and P.K. Schmieder, 1978. Water Research. 12:973-977.

Spacie, A. et al. 1983. Uptake, depuration, and biotransformation of anthracene and benzo[a]pyrene in bluegill sunfish. Ecotoxicology and Environmental Safety. 7:330-341.

Stegman, J.J. and J.M. Teal. 1973. Marine Biology. 22:37-44.

Tay K.L. et al. 1992. Sediment bioassessment in Halifax Harbour. Environmental Toxicology and Chemistry. 11:1567-1581.

Verasani, U. et al. 1978. Toxicol. Appl. Pharmacol. 44:277-289.

Villholth, K.G. 1999. Colloid characterization and colloidal phase partitioning of polycyclic aromatic hydrocarbons in two creosote-contaminated aquifers in Denmark. Environ. Sci. Technol. 33:691-699.

Wan, M.T. 1994. Utility right-of-way contaminants: polycyclic aromatic hydrocarbons. J. Environ. Quality. 23:1297-1304.

Webb, D.A. No date available. Creosote, its use as a wood preservative in the railroad transportation industry with environmental considerations. American Wood Preserver's Association.

Wendt, P.H. et al. 1996. Wood preservatives leachates from docks in an estuarine environment. Arch. Environ. Contamin. Toxicol. 31:24-37.

Zepp, R.G. et al. 1980. Assessing the photochemistry of organic pollutants in aquatic environments. In, Dynamics, Exposure and Hazard Assessment of Toxic Chemicals. R. Haque, Ed. Ann Arbor Sci. Publ. Inc., Ann Arbor, Michigan: 69-110.