THE USE OF PESTICIDE-TREATED WOOD PRODUCTS IN AQUATIC ENVIRONMENTS:

Guidelines to NOAA Fisheries Staff for the Endangered Species Act and Essential Fish Habitat Consultations

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Introduction

The purpose of this guidance is to assist biologists for the National Marine Fisheries Service (NMFS or NOAA Fisheries) to understand the issues relating to marine uses of pesticide-treated wood and to make consistent effect determinations for projects proposing to use these products. The use of pesticide-treated wood products in aquatic environments is a wide-spread practice developed to protect the wood from degradation by aquatic organisms capable of consuming wood. This guidance briefly discusses the contaminants of concern in these products (copper and polycyclic aromatic hydrocarbons (PAHs)) and their potential to leach into the aquatic environment. This guidance also outlines Best Management Practices (BMPs) to prevent or minimize exposure of NOAA trust resources to these contaminants and describes several potential exposure scenarios that consulting biologists may routinely encounter. Of chief concern in this guidance are the effects of the contaminants on salmonids, many of which are managed under the Endangered Species Act (ESA), and the Essential Fish Habitat (EFH) provisions of the Magnuson-Stevens Fishery Conservation and Management Act (MSA). This guidance is to be used in conjunction with site-specific evaluations of other potential impacts.

The most common treatments for protecting wood used in aquatic applications contain copper or creosote (which is composed of PAHs), both of which are classified as pesticides and regulated under the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA). FIFRA requires that all pesticides be registered with the U.S. Environmental Protection Agency (EPA) through a registration process that requires a period of data collection to determine the effectiveness and hazards of the particular substance. Some pesticides such as creosote were registered with the EPA as part of the original enactment of FIFRA and were never adequately assessed for risk. The EPA has thus been "re-registering" such pesticides to ensure that their risks are fully evaluated and understood.

However, the FIFRA registration process is not in itself sufficient to address ESA concerns, and the weaknesses of the FIFRA process in this regard have been well documented in the EPA's Overview document (EPA 2004) and in a joint evaluation of the FIFRA process by NOAA Fisheries and the United States Fish and Wildlife Service (USFWS) (NMFS and USFWS 2004). In regard to copper and creosote, in 2004 and 2006 NOAA Fisheries identified specific problems with the re-registration processes for these substances and communicated those concerns in correspondence to the EPA (NMFS 2004a and 2006). Creosote was reviewed by NOAA Fisheries specifically for effects to EFH and was found likely to cause adverse impacts to such habitat (NMFS 2004a). Although NOAA Fisheries requested consultation on the re-registration of creosote, the EPA elected to postpone the re-registration process. The EPA conducted another ecological risk assessment as part of the creosote re-registration process in March 2008 (EPA 2008a and 2008b), but does not plan to initiate ESA or EFH consultations at this time.

Similarly, NOAA Fisheries commented on the EPA's ecological risk assessment for the re-registration of pesticides containing copper (NMFS 2006). In its ecological risk assessment, the EPA did not consider available studies from the large body of information on the sublethal effects of copper to critical sensory functions of salmonids (NMFS 2006); nevertheless, the EPA finalized the Re-registration Eligibility Decision (RED) for coppers despite this important omission (EPA 2006). Moreover, in March 2008, the EPA released draft Ecological Hazard and Environmental Risk Assessment documents as part of the re-registration process for inorganic arsenical wood preservatives, specifically for chromated copper arsenate (CCA), and in that assessment the EPA defers to the copper RED for consideration of impacts to ESA listed salmonids, again despite its noted deficiencies (EPA 2008c). Because these documents do not adequately consider or analyze all the impacts of these substances on NOAA trust resources, the current uses of pesticide-treated wood products cannot be assumed to be protective of those resources.

NOAA Fisheries began to grapple with the issue of pesticide-treated wood use in environments used by ESA listed salmonids in the mid-1990s as research on the topic became prevalent. In 1995, the Portland Oregon District of the U.S. Army Corps of Engineers (USACE) requested that the use of pesticide-treated wood products be covered in an ESA Section 7 biological opinion. In 1998, the Northwest Region (NWR) of NOAA Fisheries issued a position document that included a box model developed by Battell Pacific Northwest National Laboratory to predict contaminant concentrations. The NWR's position document accepted the use of pesticide-treated wood products in waters with ESA-listed salmonids, but required that project managers gather information to verify that the project would not result in unacceptable impacts to salmonids or their habitat and further required that project proposals contain some restrictions on the use of pesticide-treated wood in salmonid habitat (NMFS 1998).

These new requirements and restrictions began a long-running debate with the wood treatment industry. Over the years, as additional species of salmonids have been listed under the ESA, a significant number of new studies have been conducted; numerous agencies have issued guidelines or requirements regarding pesticide-treated wood uses; and retention standards for pesticide-treated wood products have been updated to reflect minimum levels needed for an application. In November 2004, NOAA Fisheries NWR issued a programmatic biological opinion that examined and allows the use of significant volumes of pesticide-treated wood materials (NMFS 2004b) known as SLOPES III.

The State of Washington has been renewing similar programmatic consultations with the USACE on marine uses of pesticide-treated wood products since 2005 (Sibley, Tom, pers. comm. 2008). Currently, the Washington State Department of Fish and Wildlife is developing a Habitat Conservation Plan (HCP) that will likely include BMPs and restrictions for the use of pesticide-treated wood products, when conducting activities covered under the HCP (Kreitman, Gayle pers comm. 2008). The Western Wood Preservers Institute (WWPI) put a BMP program in place, subsequently updated in 2006, to ensure that pesticide-treated wood products are produced in a manner that lessens the impact to aquatic environments and issued their own guidance document for using pesticide-treated wood products in aquatic environments. The WWPI document, Treated Wood in Aquatic Environments (WWPI 2006a), develops retention standards for various preservatives and applications, gives guidance on when individual risk assessments are needed, and discusses BMPs for the manufacture of pesticide-treated wood products, which should prevent unnecessary preservative loading and promote construction BMPs. A second document from WWPI gives more detail on pesticide-treated wood products produced using BMPs (WWPI 2006b).

After the issue surfaced in the mid-1990s, other agencies took actions related to the use of pesticide-treated wood products. The Canadian Department of Fisheries and Oceans issued guidelines in 2000 that recognize treated (and untreated) wood structures have the potential to impact the aquatic environment; however, the use of pesticide-treated wood products is still allowed in Canada with certain restrictions (Hutton and Samis 2000). The WDFW stopped the use of creosote-treated wood in freshwater lakes (Poston 2001, Kreitman, Gayle, pers. comm. 2008) and determined that other restrictions may be appropriate. The U.S. Fish and Wildlife Service's Division of Engineering produced a guidance document for wooden bridge design that requires any pesticide-treated wood be manufactured in accordance with BMPs (USFWS 2001). The USACE Los Angeles District created a standard permit condition requiring that creosote pilings be wrapped in plastic, be sealed at all joints to prevent leakage, and use rub strips or bumpers at friction points (Castanon, David pers. comm., 2004). In their Procedures for Permitting Projects guidance, the Sacramento and San Francisco Districts of the USACE determined that small boat dock, pier, and wharf construction projects that coat pesticide-treated wood materials with impact-resistant, biologically inert material will "Not Adversely Affect Selected Listed Species in California" (USACE 2006).

Acting at the request of San Francisco Bay area stakeholders in the last quarter of Fiscal Year 2004, the Habitat Conservation Division of NOAA Fisheries' Southwest Region (SWR) decided to commission an independent, third party review of the pesticide-treated wood issue. The review was conducted by Stratus Environmental Consulting, Inc (Stratus), to examine the latest data and explore guideline development that reflects the risk of using these products in environments supporting listed salmonids, as well as EFH. Additional funding support was provided by the NWR and Alaska Region Habitat Conservation Divisions. Two reports were prepared: one covering copper-based treatments, focusing on the most prevalent treatments for in-water use: ammoniacal copper zinc arsenate (ACZA) and chromated copper arsenate (CCA) (Stratus 2006a); and one covering creosote-treated wood (Stratus 2006b). These reports were generated by a comprehensive (but not exhaustive) literature review, as well as discussions with NOAA Fisheries and industry. A database containing 523 literature/reference entries was submitted by Stratus to NOAA Fisheries with the final reports. The final reports (Stratus 2006a, 2006b) were peer-reviewed through established NOAA Fisheries protocols and received public review and comment. Some changes resulted from these two review processes, and the completed final reports were submitted to NOAA Fisheries in 2007.

Stratus found that the requirements of various agencies and the recommendations of the pesticide-treated wood industry are in close alignment with one another. Both agency and industry documents note that pesticide-treated wood products can be used safely, but can cause harm if used improperly. Site-specific conditions at the project are very important in influencing and evaluating project impacts. Conditions that need to be considered include background concentrations, density of product installation, location of other pesticide-treated wood structures, and environmental conditions. Among the environmental conditions that need evaluation are flows, sediment composition, sediment oxygen levels and tidal exchange. The simple box model put forward in 1998 and the models subsequently developed by industry were determined to be generally useful for predicting levels of environmental contamination that may result from pesticide-treated wood use. The results from these models are subject to some significant uncertainties, however, and are not sufficiently developed for making precise predictions. The models should therefore be used in conjunction with site-specific information (Stratus 2006a).

The Stratus reports concluded that properly selected pesticide-treated wood can be used safely in many well planned projects, but also recommended that individual screening level assessments be conducted when the use of these products is proposed in habitat supporting ESA-listed salmonids or other anadromous species. This level of risk assessment is analogous to the initial review to determine if a project may affect, but is not likely to adversely affect, a listed species or adversely impact EFH. Levels of risk that are acceptable may be clarified at a regional scale using a programmatic biological opinion, a habitat conservation plan, or guidance. If a project passes this screening level assessment, then a more detailed site-specific risk assessment will not be required. A determination that listed species may be affected requires additional information and a site-specific risk assessment analysis. The threshold for a "not likely to adversely affect" determination requires that impacts to the listed species cannot be meaningfully measured, detected, or evaluated, and that the impact does not prevent designated critical habitat from supporting the recovery of the listed species. An EFH analysis is required to determine if the action will adversely affect the quantity or quality of the habitat. The use of pesticide-treated wood is just one aspect of a project that will be considered in these evaluations (that is, the construction and presence of the project will be evaluated regardless of the construction material for effects).

A level of local knowledge is required to make a defendable determination for these types of projects (NMFS 2003, Poston 2001); therefore, the guidance provided in this document cannot replace local review. This local knowledge can be applied by the biologist on a site-by-site basis, or will be applied by NOAA Fisheries in a programmatic opinion or regional guidance. Local knowledge is critical in determining if a proposed project can be approved informally, without a full risk assessment or biological opinion under the ESA. Documents from NOAA Fisheries, other agencies, and the pesticide-treated wood industry all recognize that individual risk assessments may need to be conducted for some projects, such as those involving installation of numerous pilings, large surface areas (e.g. bulkheads), areas with elevated background levels of contaminants, areas with other pesticide-treated wood infrastructure, areas with little flow or tidal exchange, or other especially sensitive areas (NMFS 1998, 2003, and 2004; Hutton and Samis 2000; USACE 2006; WWPI 2006a). The general acknowledgement that a closer examination may be needed is a good example of how viewpoints on this subject have come closer together in the past several years.

The use of pesticide-treated wood has declined from historical highs. Because concrete and steel have greater load bearing capacity, few wharves and other large structures are currently being constructed on wood pilings (Brooks 2003). In particular, concrete pilings are cost-competitive with pesticide-treated wood pilings over the long-term and are competing in these markets (Stratus 2006a). Most projects using pesticide-treated wood pilings (such as personal use docks), are small and involve two to five pilings spaced at least four meters apart (Brooks 2003). Many bridges are small enough to use pesticide-treated wood for decking, but the abutments are often concrete or steel and are constructed outside of the 100-year flood zone (Hayward, Dennis, pers comm., 2006). Other significant uses of pesticide-treated wood (such as railroad ties, highway sign posts and utility poles) generally only interact with the aquatic environment during flooding or through improper disposal of construction or demolition debris. Yet, despite the overall decline in pesticide-treated wood use, many instances continue to arise where project proponents wish to use pesticide-treated wood.

Contaminants

In pesticide-treated wood products, the main active ingredients of concern for effects to fishery resources are copper, in metal treated wood products, and polycyclic aromatic hydrocarbons (PAHs), in creosote treated wood. This section provides brief discussions of the toxicological concerns of these pollutants to fishery resources. The Stratus reports (2006a, 2006b) and a recent technical memorandum from the NMFS Northwest Fishery Science Center and Office of Protected Resources (Hecht *et al.* 2007) should be consulted for more details. They contain many other useful references and resources.

Copper Toxicity in the Water Column

The two main types of metal treated wood, used in aquatic applications along the west coast (including Alaska) are: ACZA and CCA. ACZA contains copper, zinc, chromium and arsenic. ACZA use is more prevalent on the west coast, because it effectively treats Douglas fir, which grows along the west coast. CCA contains chromium and arsenic, in addition to copper. Copper is the focal point of this examination, because it leaches from pesticide-treated wood products at rates that can affect aquatic resources. Copper is a common contaminant in salmon habitat, where it is deposited by mines, urban stormwater runoff, pesticide-treated wood leachate, diffusion from boat hull coatings and from algicides used in waterways or fungicides applied to cropland (WWPI 1996, Weis and Weis 1996, Baldwin *et al.* 2003, Weis and Weis 2004). Understanding the levels and forms of copper (or any contaminant) already in a system is crucial to determining the potential impact of adding more copper and will be discussed shortly, along with the benchmark concentration analysis from Hecht *et al.* (2007).

Copper leaches from treated wood products in a dissolved state. Once in the aquatic system, it can rapidly bind to organic and inorganic materials in suspension. The adsorbed material may then settle and become incorporated into the sediments. Although copper may stay bound in sediments, it may also be resuspended, dissolved in interstitial water or reenter the water column depending upon biotic, physical, and chemical conditions at the site. This copper may be taken up by organisms that inhabit or ingest benthic sediments. Additionally, the copper could be taken up by some species of plants or algae and reintroduced to the ecosystem via consumption or decomposition of these plants (Weis and Weis 2002, 1992).

Eisler (2000), Stratus (2006a) and Hecht *et al.* (2007) present usable summaries of the impacts of copper to fish and aquatic invertebrates. The breadth of this information is too long to present in significant detail in this document. Environmentally realistic concentrations of free copper are noted to impact the resistance of fishes to disease, cause hyperactivity, impair respiration, disrupt osmoregulation or impact olfactory performance. This last impact will be discussed in some detail because it is caused by lower concentrations of dissolved copper than the other impacts noted here and has implications for ESA listed salmonids. Stratus (2006a) and NMFS (Hecht *et al.* 2007) note that species mean acute 96-hour LC_{50} (lethal concentration to 50% of the test subjects) values of 19-108.1 µg Cu/l (parts per billion) were found for species of *Oncorhynchus* in freshwater environments. Although there are few studies of salmonids in estuarine conditions, the available information indicates that acute copper toxicity (i.e. mortality) typically decreases with increasing salinity (Eisler 2000, Stratus 2006a). Increasing hardness and/or salinity impairs the transport of dissolved copper across the gill membrane and thus affect toxic responses mediated by this mechanism (such as acute mortality). Olfaction mediated behavioral impacts from dissolved copper do not seem to be reduced by increasing hardness of freshwater. This is probably because the olfactory rosette is in direct contact with the aquatic environment and is not protected by a membrane. There is a distinct lack of information in the marine environment regarding potential impairment of olfaction by dissolved copper or other metals (Hecht *et al.* 2007).

A NOAA Technical Memorandum (Hecht *et al.* 2007) on the effects of dissolved copper (dCu) on juvenile salmonid sensory systems was published in October 2007. The purpose of the paper was to summarize information on effects, conduct a benchmark concentration analysis (to generate effect thresholds), and to discuss site-specific considerations of sensory system effects. A large body of scientific literature has shown that fish behaviors can be disrupted at concentrations of dissolved copper that are at, or slightly above, ambient concentrations (i.e. background) (Sandahl *et al.* 2007, 2004, Baldwin *et al*. 2003, Hanson *et al.* 1999a, 1999b). Hecht *et al.* (2007) defined background as surface waters (freshwater) with less than 3 ug/L dCu. This definition was used because the experimental water had background dCu concentrations as high as 3 ug/L dCu. Sensory system effects are generally among the more sensitive fish responses and underlie important behaviors involved in growth, reproduction, and ultimately survival (i.e. predator avoidance). Recent experiments on the sensory systems and corresponding behavior of juvenile salmonids contribute to more than four decades of research showing that dCu is neurotoxic, and directly damages the sensory capabilities of salmonids at low concentrations. These effects can become manifest over a period of minutes or hours and can persist for weeks.

Hecht *et al.* (2007) calculated benchmark concentrations (BMC) for dCu using an EPA methodology, to provide examples of effect thresholds to assist in evaluating effects of activities causing copper inputs to surface waters. BMC's ranged from 0.79 – 2.1 µg/l, corresponding to reductions in olfactory sensitivity of approximately 29.3 – 57% (95% confidence interval). The BMC examples represent the dCu concentration above background (where background is less than or equal to 3 ug/L), that is expected to affect juvenile salmonids' ability to avoid predators in fresh water. These concentration thresholds for juvenile salmonid sensory and behavioral responses fall within the range of other sublethal endpoints affected by dCu such as behavior, growth, and primary production, 0.75-2.5 µg/L. For example, Hansen *et al.* (1999a and 1999b) found that salmon will actively avoid dCu at levels 2 ug/L above background, if their olfactory abilities are not yet impaired. Sandahl *et al.* (2007) found that a three hour exposure to the same concentration of dCu (2 ug/L above background) reduced odor-evoked predator avoidance response in juvenile coho salmon that is triggered through the olfactory system.

Olfactory function becomes impaired if salmon are unable to avoid copper pollution within the first few minutes of exposure. If copper levels subsequently exceed a threshold for sensory cell death, it may take weeks before the functional properties of the olfactory system recover (Baldwin *et al.* 2003). Even transient exposure, lasting just a few minutes to copper at levels typical for surface waters from urban and agricultural watersheds, and within the U.S. Environmental Agency water quality criterion for copper, will cause greater than 50% loss of sensory capacity among resident coho in freshwater habitats (Baldwin *et al.* 2003). While that loss may be at least partially reversible, longer exposures (lasting hours) have caused cell death in the olfactory receptor neurons of other salmonid species (Julliard *et al.* 1996, Hansen *et al.* 1999b, Moran *et al.* 1992). Olfactory cues convey important information about habitat quality, predators, mates, and the animal's natal stream, thus substantial copper-induced loss of olfactory capacity will likely impair behaviors essential for the survival or reproductive success of salmon and steelhead (Baldwin *et al.* 2003).

In summary, dissolved copper (such as that leached from copper-treated wood products) has been determined to decrease salmonid olfactory performance at concentrations as low as 0.79 ug/L above background (95% confidence interval) (Hecht *et al.* 2007). The effect level may be even lower, depending upon the level of confidence selected (e.g. at a 90% confidence interval, the effects level was determined to be 0.59 µg/L above background). The severity of the impact is dependent upon the length of the exposure, as well as the concentration. Behavioral avoidance has been shown at ~2 ug/L dCu above background levels. This information should be used by resource managers and other decision makers to determine if pesticide-treated wood products can generate these elevated levels of dissolved copper, for how long, and over how extensive an area.

<u>Copper Toxicity in the Sediments</u>

For many species, the greatest probability of adverse effects is from long-term accumulation of copper in sediments. This affects prey sources through contamination, or reducing availability (NMFS 2003). Metals leached into sediments near CCA-treated wood in aquatic environments have been found to accumulate in benthic and epibenthic organisms (Weis and Weis 2004). Other animals can acquire elevated levels of copper indirectly through trophic transfer, and may exhibit toxic effects at the cellular level (DNA damage), tissue level (pathology), organism level (reduced growth, altered behavior and mortality) and community level (reduced abundance, reduced species richness, and reduced diversity) (Weis *et al.* 1998, Weis and Weis 2004, Eisler 2000). Trophic transfer from invertebrates to vertebrates is less clear than from algae to invertebrates, or from one invertebrate to another. Weis and Weis (1992, 1993, 2004) found trophic transfer from algae to snails and from oysters to predatory snails. However, researchers found no evidence of trophic transfer from amphipods to fish or higher vertebrates, even in areas with higher contaminant levels that may have impacted species richness (Weis *et al.* in Kelty and Bliven 2003). An experiment conducted by Saward *et al.* (1975 in Eisler 2000) found reduced growth in a juvenile marine flatfish species (plaice), following the food chain transfer of copper from phytoplankton to clam to flatfish. Dietary tolerance in fish may be highly species specific.

Effects of copper on invertebrates are more severe in poorly flushed areas and in areas where the pesticide-treated wood is relatively new. However, effects decrease after the wood has leached a few months (Weis and Weis 2004). Weis and Weis (2004) determined that concentrations of copper in sediments near dock pilings, in moderately flushed areas, did not show accumulation of metals. This presumably occurred because the pilings have less exposed surface area for leaching than the bulkheads. Under current industry guidelines (WWPI 2006a), site-specific evaluations are recommended (for more robust examination of effects) for projects proposing to use large volumes of pesticide-treated wood (such as bulkheads).

The threshold level of copper in sediments that begins to affect habitat functions is variable. Toxicity of sediments is mediated by the presence of acid volatile sulfides and organic matter, which bind metals and greatly reduce their toxicity, as well as by the sensitivity of local species. To consider these factors at site evaluations, project proponents should conduct tests to quantify sulfide, organic matter, pH and redox potential of the sediments to determine the effect on expected copper toxicity. Simple toxicity testing (i.e. amphipod testing) would also help determine if existing conditions are already impacting the quality of the habitat.

There are several common metrics for examining contaminant levels in sediments to determine impacts on aquatic life. Among the most common metrics are two that were generated by NOAA: the effects range low (ER-L), and the effects range median (ER-M) (NOAA 1999). These values were generated from a literature review conducted by the NOAA Status and Trends Program (Long and Morgan 1990) and subsequently updated (Long *et al.* 1995). The ER-L is defined as the 10^{th} percentile of the effects database (from this literature review) for each substance examined, while the ER-M is set at the 50^{th} percentile. Therefore, the ER-L is meant to indicate concentrations below which adverse effects rarely occur, while the ER-M is meant to be representative of concentrations above which effects frequently occur. For copper in sediment, the ER-L was determined to be 34 mg/kg dry weight and the ER-M was determined to be 270

mg/kg dry weight. The data presented in NOAA (1999) shows a percent incidence of effects at concentrations of copper <ER-L of 9.4% and of 83.7% for concentrations >ER-M. For studies with sediment copper values between the ER-L and the ER-M, the percent incidence of effects was 29.1%. A tabular presentation of ER-Ls and ER-Ms is available for downloading at:
response.restoration.noaa.gov/cpr/sediment/squirt/squirt.htm

NMFS (1998) used the ER-L copper threshold level as the effects level threshold. The area examined (lower Columbia River) had a background concentration of 20 mg/kg. This decision to use the ER-L was questioned by the pesticide-treated wood industry as being overly protective, since the NOAA database showed this level of copper in the sediments had only a 10% chance of causing an effect to EFH in the study area. NOAA agreed because of the variability observed in the background concentrations, but determined that keeping an increase of sediment copper to less than 50% is a reasonable management objective for projects (NMFS 2001).

Other sediment quality guidelines have been developed, and a significant debate continues regarding their appropriateness. The threshold effects level (TEL) guideline is defined as the 15th percentile concentration of the toxics effects data set and the median of the no-effect data set. The TEL represents the concentration below which adverse effects are expected to occur (only rarely) and is 18.7 mg/kg for copper (Buchman 1999). The probable effects level (PEL) is the level above which adverse effects are frequently expected. For copper, the PEL is 108.2 mg/kg. The PEL represents the geometric mean of the 50% of impacted, toxic samples and the 85% of the non-impacted samples (Buchman 1999). The TEL and the PEL are used in Florida and Canada. California has proposed the use of a multiple lines of evidence sediment quality guideline system, which requires sediment chemistry measurements, sediment toxicity testing, and measurements of benthic community condition (SWRCB 2008). This system is designed to eliminate uncertainties that typically remain when only sediment chemistry data is collected.

PAH Toxicity in the Water Column
Creosote is a distillate of coal tar and is a variable mixture of 200-250 compounds consisting of simple PAHs, multi-aromatic fused rings, cyclic nitrogen-containing heteronuclear compounds and phenolic substances (EPA 2008a, 2008b, 2003). PAHs are the dominant class of compounds found in creosote and comprise approximately 85% of creosote's mass (EPA 2003, Stratus 2006b). Sixteen of the top 17 PAHs found most commonly in creosote are on the EPA's List of Priority Pollutants, pursuant to the Clean Water Act (EPA 2003). Currently, there are two formulations of creosote in use: P1/P13 (used for aquatic applications) and P2 (used for railroad ties and utility poles) (Stratus 2006b).

PAHs are released from wood treated with creosote and are known to cause cancer, reproductive anomalies, and immune dysfunction; to impair growth and development; and to cause other impairments in fish exposed to sufficiently high concentrations over periods of time (Johnson *et al.* 1999, Karrow et al. 1999, Johnson 2000, Stehr *et al.* 2000, Collier *et al.* 2002, Johnson *et al.* 2002, Sherry et al. 2005, Stratus 2006b). When exposed to creosote-derived PAH concentrations as low as 320 µg/L, Spot (*Leiostomus xanthurus*, also called flat croaker) exhibited fin erosion and epidermal lesions (Sved *et al.* 1992). Embyonic exposures can result in edema (swelling) of the yolk sack, hemorrhaging, disruption of cardiac function, enzyme induction, mutation of progeny, craniofacial and spinal deformities, neuronal cell death, anemia, reduced growth and impaired swimming (Barron *et al.* 2003, Billiard *et al.* 1999, 2002, Brinkworth *et al.* 2003, Marty *et al.* 1997: all cited in Barron *et al.* 2004, Incardona *et al.* 2004, 2005, Wassenberg and Di Giulio 2004a, 2004b). Exposure to sunlight has been observed to result in a 48-fold increase in toxicity of some PAHs to herring larvae (Barron *et al.* 2003), an increased medaka embryo failure rate (Diamond et al. 2006), impacts to invertebrates (Pelletier *et al.* 1997, Swartz et al. 1997) and resulted in as little as 2 µg/L becoming toxic to calanoid copepods (Duesterloh *et al.* 2002)). Impacts to phytoplankton and zooplankton communities have also been reported in the literature (Sibley *et al.* 2004, 2001a, 2001b, Bestari *et al.* 1998a). Traditional LC_{50} testing (24 and 96 hour tests) for invertebrates and fish resulted in a range of effect levels generally between 0.1 and 4 mg/L (parts per million). This concentration is greater than what is typically encountered, even in polluted surface waters (Eisler 2000). Crustaceans tend to be relatively more sensitive in these tests than fish species.

Several studies demonstrate that PAHs harm the egg-larval lifestage of Pacific herring (Vines *et al.* 2000, Carls *et al.* 1999), surf smelt (Misitano *et al.* 1994) and pink salmon (Heintz *et al.* 1999, Bue *et al.* 1998). Vines *et al.* (2000) studied the hatching success of herring eggs (exposed to PAHs leaching from creosote pilings) and found that 0% of the eggs attached to the piece of piling hatched. In addition, 40-50% of nonattached eggs had delayed development and the surviving embryos all had morphological abnormalities. This study established a LC_{50} for creosote leachate of 50 µg/L for the herring embryos, with sublethal effects observed at concentrations as low as 3 µg/L. The applicability of this study, to actual environmental conditions, is weakened by its static water design with small chambers. An additional field observation of toxicity was made by the same author (Cherr and Vines 1997), but mortality may have been affected by other factors (such as temperature and salinity). The creosote formulation and loading at this observation was not reported. Carls *et al.* (1999) showed that total dissolved PAH concentrations from weathered oil of 0.7 µg/L caused morphological malformations, genetic damage, inhibited swimming, decreased size and mortality of larval Pacific herring. Sublethal effects (such as yolk sac edema and delayed mortality) were observed at concentrations as low as 0.4 µg/L total dissolved PAH. Poston (2001) reviews several other studies of the effects of weathered crude oil (high molecular weight PAHs such as those found in creosote contaminated sediments) and other PAHs or sources on various endpoints including the spawning success of pink salmon and herring.

PAHs bioaccumulate in many invertebrate species (Varanasi *et al.* 1989, 1992; Meador *et al.* 1995), but are metabolized significantly by many vertebrates (including fishes) where they are converted to water-soluble forms and excreted (Varanasi *et al.* 1989, Stratus 2006b). Some of the intermediate metabolites in this process exhibit carcinogenic, mutagenic and cytotoxic properties. Metabolic capacity is generally very high in vertebrates, intermediate in crustaceans and limited in bivalves (Meador *et al.* 1995).

PAH Toxicity in the Sediments

The main exposure scenario of concern for PAHs, including those leached from creosote treated wood, occurs as they accumulate in sediments and are assimilated into the food web. As mentioned in the last section, the concentration of creosote derived PAHs required to cause acute mortality to nonlarval fish is generally high enough that the level rarely occurs. More frequently, it is the chronic and dietary exposures to the higher weight PAHs remaining in sediments that cause the effects listed in the last paragraph (i.e. cancer, reproductive anomalies, immune dysfunction, growth and development impairment, and other impairments to fish exposed to sufficiently high concentrations over periods of time or exposed during their egg or larval life stages). Effects are more prevalent in benthic species (such as English sole, winter flounder and brown bullhead), due to their frequent contact with the sediments (Stratus 2006b).

There is a significant debate over what level of PAHs in sediments cause the adverse effects discussed. Research by scientists at the NMFS Northwest Fisheries Science Center (Johnson *et al.* 2002) suggested a sediment threshold level for total PAH of 1 part per million (mg/kg dry weight) would protect estuarine, bottom dwelling fish (such as the English sole examined in the study), from detrimental effects (such as liver lesions, spawning inhibition and reduced egg viability). This level (1 mg/kg) was the lowest at which effects to English sole began to be observed. A model developed as part of this study predicted a 10-fold increase in DNA adducts (a complex formed when a carcinogen combines with DNA or a protein) at 5 mg/kg total PAH compared to control fish, resulting in liver disease to approximately 30% of the exposed fish and increasing failure to spawn. The authors noted a concern that other carcinogenic contaminants (PCBs, chlorinated pesticides, and trace metals) that were present in the sediments of the Puget Sound at the various study locations may be significant confounding factors.

The ER-L for total PAHs is approximately 4mg/kg, while the TEL (approximately 1.7 mg/kg) is closer to the level suggested by Johnson *et al.* (2002). The concentrations of concern are even lower for total high molecular weight (HMW) PAHs, which typically remain in the sediments, with an ER-L of 1.7 mg/kg and a TEL of 0.66 mg/kg (Buchman 1999). These are environmentally realistic concentrations that may be exceeded in industrialized or urbanized areas; however, these are the levels where effects are predicted to begin. The ER-M for total PAHs is approximately 44.8 mg/kg (total HMW PAH = 9.6 mg/kg), while the PEL is approximately 16.7 mg/kg (total HMW PAHs = 6.7 mg/kg). Sediments with PAH levels above the lower thresholds warrant protection from additional contamination in order to protect the function of the sediment for EFH as well as ESA listed species. The next discussion of this information considers if pesticide-treated wood products can generate these elevated levels of PAHs in sediments, for how long and over what area of a waterbody.

Leaching Rates, Subsequent Exposure and Model Evaluation

Now that the toxicity of the contaminants has been summarized, it must be determined if the use of pesticide-treated wood products will result in conditions that may impact NOAA trust resources in a manner which can be meaningfully measured, detected, or evaluated. Conditions should be evaluated for adverse affects to species listed under the ESA, limitations to the functions of designated critical habitat to support recovery of a listed species, or adverse affects to the quantity or quality of EFH. This section will briefly present information on leaching rates of pesticide-treated wood products, models developed to predict leaching rates, subsequent concentrations in sediments and water, and studies of pesticide-treated wood installations. Much more detail is available in the Stratus reports (2006a, 2006b) and their source documents.

Copper Treated Wood – ACZA and CCA

These two copper-based formulations are often used for in-water portions of structures. There are several other prominent formulations of wood treatments containing copper that may be used in over-water applications. These formulations were not examined in significant detail during the Stratus (2006a) literature review, but concerns about their use in over-water structures is considered with ACZA and CCA later in this document.

The leaching of copper from ACZA and CCA treated wood demonstrates a general trend of higher initial leaching rates that decrease rapidly within days. Within a few weeks to months, copper leaching decreases to very small levels; however, this is dependent upon pH, temperature and other variables. ACZA leaching rates in freshwater were very low within 10 days of installation (Brooks 1995b, NMFS 1998, Hutton and Samis 2000). The majority of leaching occurred from CCA pilings during the first 30-90 days after installation (Brooks 1995a, Weis *et al.* 1991, NMFS 1998, Hutton and Samis 2000, Kelty and Blevin 2003). ACZA leaches a greater total mass of copper in freshwater compared to CCA, but ACZA leaches for a shorter time.

The spreadsheet models reviewed by Stratus (2006a) are available for public use on-line at the WWPI website: www.WWPInstitute.org. Models for ACZA (ACZA.xls) and CCA (CCAPRISK.xls) were accessed on December 7, 2007 and results are presented in Table 1. The ACZA model gives both fresh and marine water outputs simultaneously. All other inputs in the model that produced Table 2 were left at their default settings.

Table 1. Leaching Rates (μg/cm^2/day) of a single 15 cm diameter piling, Freshwater salinity 0 ppt, Saltwater salinity 30 ppt

Time		Day 0	1	3	10	30	90
ACZA	Freshwater.	118.79	77.35	32.8	1.63	0.00	0.00
	Saltwater	32.55	10.68	1.15	0.00	0.00	0.00
CCA	Freshwater.	1.39	1.33	1.20	0.86	0.33	0.02
	Saltwater	2.86	2.72	2.47	1.77	0.68	0.04

Table 1 is intended only to illustrate the rapid decline in leaching rates. Project specific information (e.g. piling diameter, tidal velocities, water depth, etc.) can be entered into the on-line spreadsheets to generate more specific predictions of leaching rates.

Problematic exposure scenarios need to be evaluated along with leaching rates. Review of projects affecting juvenile salmonid habitat must consider if problematic concentrations of copper will leach into habitat when salmonids are present. In well mixed areas, dilution is often sufficient to decrease the concentration of CCA or ACZA to inconsequential levels. In other circumstances, the project could be scheduled to allow leaching when salmonids are less likely to be present. The model presentation used in NMFS 1998 (Table 1 in that position paper - acknowledging uncertainties discussed later in this document), shows that installation in an area with a current velocity of 10 cm/sec or more, does not increase water column concentrations of copper more than 0.43 µg/L for 100 or fewer pilings. This applies even in the smallest, most dense cross-sectional area evaluation. For reference, the lowest benchmark concentration calculated by NMFS (Hecht *et al.* 2007), at the 95% confidence interval, for impacts to salmonid olfactory performance was an increase of 0.79 µg/L. NMFS (1998) shows increases of less than 0.79 µg/L are predicted for other projects with greater numbers of piling at this current velocity. However, exceedances of this benchmark typically occur in only the densest installations. There are also multiple regression equations presented in the position document (NMFS 1998) that can be used to generate predictions of copper concentrations for individual projects of different sizes.

The majority of projects proposing to use pesticide-treated wood are smaller than the 100 piling size (at any installation density) which predicts potentially problematic water column concentrations at current velocities of 1 cm/sec or less. Because of the superior weight bearing properties of concrete and steel, most projects currently proposing to use pesticide-treated wood are small-scale (such as personal use docks and rural bridges). These projects typically have between two to five piling bents spaced at least four meters apart (Brooks 2003). Some of these projects use pesticide-treated wood mainly in the decking.

The Poston model utilized in NMFS (1998) and another spreadsheet based model developed by Brooks (1997b) were reviewed in Stratus (2006a). Although there is variability between the predictions of the leaching models and the observed leaching rates in laboratory studies, Stratus determined that there is little or no bias in leaching model under or overpredictions. The subsequent environmental models, which are used to predict water column concentrations and impacts to sediments, appear to capture the overall trends in leaching rates reasonably well. However, the review cautions that site specific conditions need to be known to provide useful predictions (Stratus 2006a). The remaining uncertainty can be addressed in a large-scale or regional guidance through other mechanisms (such as limiting the number of pilings that a project may install), without going through a more thorough analysis. NMFS (2004b) did this in the lower Columbia River and Oregon by essentially pre-approving projects that propose 50 or less copper treated pilings. This effectively serves as a margin of safety from the uncertainties in the 100 pile predictions. Other requirements, such as limiting the width of the structure, address other project related impacts. This level of pre-approvals still captures a large percentage of proposed projects, making it an efficient solution from both regulatory workload and species protection points of view.

NMFS (1998) also presented estimates of total sediment copper concentrations over a variety of project cross sectional areas and current velocities, based upon the Poston box model used by the USACE in their biological assessment (USACE 1996). This is presented as Table 2 in the NMFS (1998) document. A background sediment copper concentration of 20 mg/kg was assumed in the model and this must be adjusted by substituting local background concentrations for the default value (if using the model predictions as a guide). Significantly elevated concentrations are predicted to occur near the pilings, but elevated levels further away from the pilings are negligible. This finding is similar to the Washington State White Paper (Poston 2001), which found that increases in sediment metal concentrations were limited to within 10 feet from small pesticide-treated wood structures in marine and freshwater habitats. Weis and Weis (1996) and Weis *et al.* (1998) measured increases of copper in sediments adjacent to bulkheads (<1m) constructed of CCA treated wood. Additionally, Weis and Weis (1994) examined a number of dock sites (rather than bulkheads) and found that these sites did not have increased metals in sediments adjacent to pilings or any consistent differences in benthic communities. The authors concluded that leachates from pilings in reasonably well-flushed areas do not have negative effects in the immediate vicinity. This is of importance because industry guidelines (WWPI 2006a) state that high surface area or high density uses of pesticide-treated wood products (such as bulkheads) should undergo site-specific assessments. Applicants can be requested to follow industry guidelines by generating necessary background information and monitoring, which is needed to make accurate effects determinations. Stratus (2006a) concluded that metals leaching from pesticide-treated wood structures resulted in only minor accumulation over a limited area (within several meters) in well mixed waters. The resulting accumulations have not been associated with significant biological effects, except in close proximity to the structures (usually bulkheads that also affect the biological community by increasing scour of fine sediments away from the structure due to reflected wave action).

<u>Creosote Treated Wood</u>
Leaching and deposition from creosote-treated wood products were also evaluated in the position paper (NMFS 1998). A number of creosote formulations were used in the past, which complicates the evaluation of leaching studies (Stratus 2006b). There are also fewer studies with modern creosote-treated products than copper-treated products, which results in additional uncertainty when making effects determinations for new projects. Like CCA and ACZA, the most rapid leaching of PAHs from creosote-treated products occurs initially (Sibley *et al.* 2004, Bestari *et al.* 1998b, Ingram *et al.* 1982); however, detectable leaching occurs for years and perhaps much longer (Stratus 2006b). The spreadsheet model available through WWPI, and reviewed by Stratus, uses an age input in years (rather than days). The USACE (1997) biological assessment recognized that leaching of PAHs occurs for years after the installation of pilings and referenced a study by Ingram *et al.* (1982) that determined rates for a 12 year-old pile (8.0 $\mu g/cm^2/day$). NMFS (1998) recognized that the sediment PAH model used by the USACE did not account for resuspension, turbulence, lateral dispersion or biodegradation of PAH compounds over time and may overestimate the accumulation of PAHs in the sediment over the long-term.

Brooks *et al.* (2006) is the latest publication documenting the results of a long-term study conducted with Fisheries and Oceans Canada in the Sooke Basin, British Columbia, to address the longer term PAH leaching from creosote. The study included the construction of three dolphins: each containing six pilings. The study included: untreated wood, non-BMP produced wood (that had been installed elsewhere for 8 years before being moved to this location) and BMP produced piling that was over-treated (27 pounds of product per cubic feet compared to the AWPA standard for marine use of 16-20 pounds of product per cubic foot). The Sooke Basin study site had slow currents (2.3 cm/sec at the surface and < 2 cm/sec at depth (~12 meters)), and low baseline sediment concentrations of PAH, making it a good location to conduct the study. During the first year, creosote[1] accumulated in the sediments, within 7.5 meters of the BMP treated structures at concentrations of <7 to 10 mg/kg TPAH. By the end of the ten year study period (Brooks *et al.* 2006), elevated sediment levels were detected only within 2.5 m from the BMP pilings (5.2 mg/kg TPAH in the upper 3 cm of sediment and declining at 21 cm to 0.140 mg/kg TPAH). The creosote contamination at the 2.5 m distance was not uniform. The authors reported that the initial TPAH release did not seem to be toxic to the local infaunal community (Goyette and Brooks 1998). Toxicity was observed in laboratory tested samples on standard organisms, for sediments within 2 feet of the treated pilings (Goyette and Brooks 1998, 2001). Water column concentrations were not measured at this time.

<u>Exposure from Over-water Use of Pesticide-treated Wood Products</u>
Significant quantities of pesticide-treated wood products are used in above-water structures and decking which warrant examination and sound management recommendations (WWPI 2006a). Pesticide-treated wood structures placed in or over flowing waters will leach copper and a variety of other toxic compounds directly into the stream (Weis and Weis 1996, Hingston *et al.* 2001, Poston 2001, NOAA 2003). These structures can be sources of copper to waterbodies from leaching during rain storms or washing, splashing, from abrasion caused by foot or vehicle traffic, or release of sawdust or other wastes during construction or maintenance procedures. Creosote-treated products can release PAHs from these same mechanisms and from exposure to the sun. Sunny, warm conditions cause creosote to be more mobile and “ooze” or blister out of the product. The droplets can then be released to the waterbody.

Exposures of this type are often sporadic and can occur for a longer period of time because the pesticides do not have a chance to be removed rapidly, except in areas of high precipitation. Weathering is based largely upon rain intensity and duration and is thought to mainly occur during the first year, especially in areas which experience regular rainfall (Brooks 1997b). Although overwater structures will release contaminants, the biologist must assess the situation to determine if the releases will result in adverse effects to NMFS trust resources. Similar to in-water structures of pesticide-treated wood, infrastructure where significant dilution will occur will have sporadic inputs of contaminants that are less likely to have impacts that are measureable, detectable or that can be meaningfully evaluated.

Studies conducted on bridges and boardwalks are useful for examining the potential contribution of contaminants from overwater structures. Two bridges constructed of

[1] Measured as TPAH

CCA treated wood were studied by Brooks (2000) in Florida. One was over freshwater and one in a marine environment. Both sites had some sediment copper contamination, though the levels were low and limited in space (approximately six feet downstream of each bridge). The marine bridge increased copper concentrations from 2.3 mg/kg (background) to 25.1 mg/kg directly below the bridge and the concentration dropped to 7.95 mg/kg at 1.5 feet downstream. Concentrations were at background levels 10 feet from the bridge. The freshwater bridge was two years old when studied. Background concentrations were 0.63 mg/kg in the sediments. They were to 2.1 mg/kg beneath the bridge and for three feet downstream, but were near background concentrations at 20 feet downstream of the bridge. Abundance and diversity of aquatic insects were measured and did not indicate impacts, while bioassays did not indicate toxicity.

A boardwalk study (FPL 2000) was conducted in Oregon on a 1,800 foot long boardwalk, constructed in a wetland area from three different copper treated products, to evaluate the product's environmental effects. The environment was slow moving freshwater, with fine grained sediments and heavy rainfall. The boardwalk was monitored for one year after construction. Copper accumulations from both CCA and ACZA formulations were found to vary temporally and spatially. The baseline levels of copper in the surface sediments varied from 17-24 mg/kg at the CCA treatment site. After two months the maximum copper concentration under the boardwalk was as high as 201 mg/kg, with a median concentration of 64 mg/kg. These levels increased further at 5.5 months, with a maximum detection of 219 mg/kg and a median level of 112 mg/kg. These levels decreased by the 11th month, with a maximum detection under the boardwalk of 115 mg/kg and a median concentration of 85 mg/kg. Copper levels were also elevated two feet from the boardwalk in the surface sediments, with a median concentration of 51 mg/kg. Copper levels in the surface sediments at the next monitoring point (five feet from the boardwalk) remained within background levels, although the data shows a gradual increase in low-level contamination moving away from the structure. This localized pattern of distribution indicates that the majority of leached copper was bound to suspended materials that settled into the sediments. Monitoring of leaching found that the greatest amount leached during initial rainfall. It must be noted that the CCA materials used in this trial were prestained. Experiments on the efficacy of coatings to minimize leaching from CCA-treated wood, found that one coat of latex primer, followed by one coat of oil-based paint or two coats of penetrating, water-repellent deck stain were both effective for reducing the leaching of copper, arsenic and chromium by more than 99% (FPL 2001a). Materials not treated in this manner may leach more copper.

At the ACZA site, baseline copper levels varied from 18-21 mg/kg in the surface sediments. Elevated levels of copper were observed as soon as 10 days after construction. At 2.5 months the median concentration under the boardwalk was 47 mg/kg, although there was a sample containing 569 mg/kg under the boardwalk, and others measuring 122 mg/kg and 226 mg/kg at one foot and five feet away from the boardwalk respectively. The distribution was more heterogeneous than the CCA treated wood site. Concentrations decreased at the six month sampling period, but increased again during the 11.5 month post construction sampling. This was likely due to a large rainfall event that flushed sediments away from the ACZA treated site. This indicates that the dissolved copper, leaching from the boardwalk, was being absorbed and settling into the sediments. At 11.5 months, the mean copper concentrations were elevated above surface sediment levels, within one foot of the structure (95% confidence interval). Concentrations were slightly elevated within 5 feet of the structure, but not at this level of statistical significance. Like the CCA treated wood in this evaluation, initial leaching rates were highest during initial exposure to rainfall, although the mass of copper leached was correlated with total rainfall.

Invertebrate sampling was evaluated to detect potential adverse affects to these communities and habitat quality. Total species richness, sample abundance, dominant sample abundance, and Shannon's and Pielou's indices were calculated. These indices did not show a significant reduction in habitat quality, and no taxa were extirpated from the study area, despite the elevated concentrations near the boardwalk. Replicate samples were not taken, with the exception of artificial substrates that allowed for expeditious sampling. These artificial substrates excluded some taxa; therefore, the sampling design does not allow for a more thorough examination. Differences in abundance and diversity near and far from the boardwalk seem to occur in some datasets, but this could simply be due to natural variability. All of the indices were comparable to the control, within 10 feet of the boardwalk.

The author noted that the high rainfall and large volume of pesticide-treated wood used in construction of the boardwalk represented a severe leaching hazard. A project of this size would likely be considered as an example of a "substantial project having large treated wood surface area," as noted in WWPI (2006a). This would warrant an individual risk assessment and therefore, would generate the data for a more thorough examination. This determination may depend upon the proposed location and potential for direct exposure of salmonids. Affects may not be detectable or measurable in some locations or may be rather limited in area as illustrated by this study.

<u>Exposure from Construction Debris</u>

If pesticide-treated wood sawdust or shavings (generated during construction) are allowed to enter soil or water below a treated structure, they make a disproportionately large contribution to environmental contamination. Impacts from the leaching of construction debris immersed in water are vastly greater than from solid wood (FPL 2001b, Lebow and Tippie 2001, Lebow *et al.* 2004). Construction debris may release 30 to 100 times more preservative than typical submerged pieces, due to the increased surface area available for leaching. Collection of construction debris should be stressed. Storing pesticide-treated wood out of contact with standing water and wet soil, as well as protecting the wood from precipitation significantly reduces the likelihood of chemical leaching during construction (Lebow and Tippie 2001, FPL 2001b).

Linkage of Toxicity, Modeling, Field Studies and Expected Impacts

Copper-treated pilings leach relatively quickly, reaching low exposure levels in a matter of days to several weeks, depending mainly on formulation. For in water uses, the highest leaching occurs in the first few days. Within this time, the resulting water column concentrations may be high enough to affect salmonid olfaction ability (an increase ≥0.79 µg/L dCu), depending upon the size of the project and the available dilution. The biologist will need to determine the likelihood of salmonid early lifestage presence and potential exposure. Sediment contamination is a possibility, but was not noted at problematic levels from dock structures, compared to bulkhead structures. The models used by the NWR of NOAA Fisheries (NMFS 1998, adapted from USACE 1997) and the industry (Brooks 1997a and 1997b) can both over and under predict leaching rates, but do capture the trends. Although they can not be used for exact predictions, the models can provide useful site-specific predictions when used with site-specific information and when BMP produced wood is installed. Overall, the models consistently overestimate water column concentrations because of simplifying assumptions (such as all the piles being installed simultaneously) and the difficulty of accounting for mixing by turbulence (Stratus 2006a). The models should perform well enough for a site-specific or regional examination to determine if a project is likely to have adverse effects on NMFS trust resources because their conservative tendencies help to offset some of the uncertainties. Most projects, which are small enough to not require a more detailed site-specific assessment, are likely small enough to not be a concern (i.e. 2-5 pilings spaced at least 4 meters apart (Brooks 2003), perhaps with some pesticide-treated wood decking).

Creosote-treated wood leaches significantly in freshwater, but has been banned from use in most freshwater areas by the West Coast states. It leaches detectable amounts of PAHs for years to decades in marine environments. The resulting sediment contamination is usually localized near the structure and diminishes over time. Elevated PAH levels in sediments have been implicated in causing tumor growth and reducing fecundity in bottom dwelling fish. Creosote-treated wood can cause these levels in sediment in some habitats over localized areas. There may be many sources of PAH in the environment, causing elevated background concentrations. Numerous waterbodies are listed as impaired, due to excessive levels of PAH. Additional contaminant loading is not advisable in these impaired waterbodies.

Water column concentrations of PAH from BMP creosote treated wood sources are not expected to cause detectable, acute effects under most exposure scenarios. Most vertebrates, including fish, can metabolize PAHs fairly efficiently. Several studies document impacts to herring eggs exposed to PAHs. The Vines *et al.* (2000) study is useful for establishing a LC_{50} for herring hatching success following creosote-derived PAH exposure. Significant uncertainty regarding whether this level of impact will be reached in the field warrants a cautious approach in areas where herring spawn and potentially in areas where the egg and larval lifestages of other species may be similarly exposed. Creosote treated piling installations in the Los Angeles area have been required by the USACE to be wrapped for many years to reduce exposure to the environment (Castanon, David pers. Comm., 2004)

Precedent has been set to permit a predetermined number of pilings to be installed without triggering full ESA and EFH consultations (NMFS 2004b). This more site specific analysis should be done for local areas, in which these types of projects are being proposed. Overwater use is significant enough to warrant the use of BMPs (e.g. construction BMPs, use of BMP treated wood), but is not likely to result in problematic concentrations except in the most sensitive environments. BMPs should be used to minimize unnecessary risk for both in-water and above-water utilizations.

Best Management Practices

The above sections show that the properly planned and executed use of pesticide-treated wood products is unlikely to cause detectable impacts to ESA listed salmonids, in many use scenarios; however, uncertainties remain in the underlying leaching experiments, field studies and modeling efforts (Stratus 2006a). Evaluation of in-service structures show that leaching rates vary by wood dimensions, wood species, treatment practices, fixation, age of the structure, type of exposure, construction and maintenance practices, and site-specific conditions (Lebow 1996, Lebow *et al.* 2004). The potential cumulative effect of these uncertainties has led numerous agencies (NMFS (1998, 2003, 2004), Fisheries and Oceans Canada (Hutton and Samis 2000), USDA (Lebow and Tippie 2001), Washington Department of Fish and Wildlife and Department of Ecology (Poston 2001), USACE (2006, Castanon, D., pers. comm. 2004)) and industry (WWPI 2006a, 2006b) to recommend BMPs to minimize avoidable and unnecessary risks to the environment. The following section on BMPs should be considered by the project proponent, the permitting agency (usually the USACE for NMFS purposes) and the reviewing agency (NMFS), and all warranted practices put into place for a project. Some districts of the USACE have standardized several of these BMPs resulting in an increased regulatory certainty for project proponents and more expedient review and approval processes.

Proper Material Selection – BMP Pesticide-treated wood

Perhaps the most important BMP is simply proper selection of pesticide-treated wood materials for a project. At the basic level, this means that the pesticide-treated wood product contains no more than the minimum level of pesticide necessary, as specified by the American Wood Preserver's Association retention standards Higher retention levels do not lead to extra durability. They only lead to increased leaching and subsequent impacts (Lebow and Tippie 2001).

The simplest way to ensure that the wood to be used has been properly treated is to require the project proponent to use products that have been BMP certified through a third party inspection process. The WWPI has set up such a procedure, so that products can be verified as being produced in compliance with production BMPs (WWPI 2006a). This means that they will be treated to proper retention standards and be processed to maximize fixation of the product. This would result in lower leaching rates (as used in the environmental exposure models). This is crucial to insuring that predicted levels of contamination are in fact those which are likely to occur. BMP pesticide-treated wood is denoted with a written certification from a company accredited by the American Lumber Standard Committee (in compliance with regulations of the U.S. Department of Commerce), or through the presence of a BMP mark as seen in the WWPI documents (2006a, 2006b). However, in the event that an improperly labeled material arrives at a job site, a visual inspection and rejection of materials (with visible residues or bleeding) requirement is still recommended.

Proper Material Selection – Environmental Conditions

Proper selection of pesticide-treated wood products is also based upon environmental conditions. Creosote-treated wood and copper-treated wood seem interchangeable in marine environments and selection of materials is often a matter of personal preference. Use of creosote-treated products is already restricted, or not recommended, in many freshwater environments in California, Oregon, Washington and Canada (Stratus 2006b, Hutton and Samis 2000). If a project proponent insists on using creosote-treated wood in spawning habitats of vulnerable species and lifestages (such as Pacific herring or pink salmon), or in a PAH impaired waterbody, then wrapping the pilings to form a physical barrier between the leachable material and the aquatic environment may be appropriate mitigation. This treatment is required by the Los Angeles District of the USACE (Castanon, David pers comm., 2004), to protect fish habitats. In areas without such restrictions, the condition of the sediments needs to be examined to determine if creosote could be used with minimal impact. Creosote-treated wood pilings should not be installed in anoxic sediments or areas with low dissolved oxygen concentrations, as the PAHs will degrade more slowly due to lack of oxygen and can accumulate to problematic levels. These are site-specific assessments where local knowledge is vital. In other less sensitive areas, it is important to remember that the long-term study of modern BMP creosote-treated wood in the field (Sooke Basin study), found limited contamination only in areas adjacent to the structures (Brooks *et al.* 2006, Goyette and Brooks 1998, 2001).

If pesticide-treated wood is used in overwater structures, BMP pesticide-treated wood can minimize effects to sensitive environments. For example, bridges or decks built over low- flow areas that support salmonids may need to be BMP treated to minimize leaching into a waterbody with minimal available dilution. This may be especially important if the structure will be cleaned during the low flow season. Cleaning and maintenance activities (such as aggressive scrubbing, power-washing, or sanding) can also remove particles of pesticide-treated wood and deposit them in soil or water beneath the structure (Lebow and Tippie 2001). Wooden bridges built without a wearing surface (so that vehicles ride directly on a pesticide-treated wood deck) may abrade because vehicle traffic wears away the preservative treatment over-time and exposes new surfaces of the wood to leaching (Brooks 2000, Ritter *et al.* 1996a and 1996b). Similarly, foot traffic will abrade pesticide-treated wood used in pedestrian bridges unless prevented by a wearing surface such as synthetic mats, coatings, metal sheets, or sacrificial plywood sheets (DeVenzio undated in NMFS 2004b, Lebow *et al.* 2003). Coatings will be discussed later in this document. Otherwise, products which leach the lowest amount of copper should be encouraged. In general, this seems to be CCA (Kennedy 2004, Stefanovic and Cooper 2004, Stratus 2006a), although the use of products with arsenic is now limited by EPA due to human health concerns in some applications.

Require Site Specific Assessments for all Larger Scale Projects

Table 1 in NMFS (1998) presents an adaption of information from the Poston box model, developed for the USACE biological assessment (USACE 1997) for the lower Columbia River. It was reviewed by Stratus (2006a) and determined to be useful in providing site-specific predictions where site-specific conditions are known. It can be used as a screening tool to predict increases in dCu above background concentrations if current velocity, the cross-sectional area of the projects, pH and the number and size of pilings to be installed is known. For copper-treated products, water column increases of <0.79 μg/L dCu were consistently predicted at current velocities of 1 cm/sec for projects with 24 or fewer pilings only at the smallest cross sectional area (i.e. cross sections ≤ 200 m^2). This may show that the pesticide-treated wood impacts of many smaller projects (such as personal boat docks) do not generally cause problematic levels of copper in the water column. If current velocities are 10 cm/sec or greater, all modeled installations up to 100 pilings were acceptable showing the importance of dilution to determining a measurable or detectable level of impact. All installations of 100 pilings or less were found to not have problematic sediment impacts (i.e. <50% increase over the background concentration of 20 mg/kg copper) at current speeds of 1 cm/sec or greater.

A key assumption to the environmental exposure model (NMFS 1998) is that all pilings are installed simultaneously, which does not accurately reflect the logistics of construction. A project that uses 100 pilings will require several days to install, under the best of conditions. That means that the water column concentrations, projected by the model for the 100 piling scenario, are conservative and are likely to be lower in reality. Multiple regression equations, found on pages 10 and 11 of NMFS 1998, can be used for projects proposing between 24 and 100 pilings if the project is not in an area where a more thorough analysis has been conducted or programmatic consultation is in place. The SLOPES opinion (NMFS 2004b), covering the State of Oregon and lower Columbia River, is an example of this and effectively pre-approved projects installing 50 pilings or less (with some other restrictions). The WWPI (2006a) recommends conducting a site-specific assessment if more than 100 pilings are proposed for a project. This recommendation can be provided to the action agency and project proponent if more information is needed to determine the potential impacts of larger projects.

<u>Timing of Installation</u>
Restricting when an action can take place is a well established method of preventing or minimizing impacts to listed species and/or sensitive habitat components. The timing of use of an area by federally managed, or ESA listed species, should be determined. If a proposed project is of sufficient scope (that it may release contaminants at problematic levels), construction timing windows may be useful. In California, there are often time periods in larger rivers, bays and estuaries when salmonids are not present and other periods when only migrating adults are present. In the Northwest Region and Alaska, many of rivers and estuaries are used extensively by incubating and rearing juvenile salmonids; therefore, there may be nearly constant juvenile emigration. Timing restrictions (sometimes called work windows), may already be in place for other activities (e.g. dredging), or for related impacts (e.g. pile driving, turbidity).

<u>Construction BMPs</u>
As noted previously, elevated contaminant releases from pesticide-treated wood materials can occur during the construction process. This is due to the high surface areas of debris (such as sawdust) and the exposure of the inner portions of the wood where the chemicals may not be as strongly fixed initially. The use of construction BMPs reduces

unnecessary risks to aquatic habitats and is recommended by numerous agencies (NMFS (2004, 2003), Fisheries and Oceans Canada (Hutton and Samis 2000), USDA (Lebow et al 2000, Lebow and Tippie 2001)), industry (WWPI 2006a, 2006b) and subject reviews (Poston 2001, Stratus 2006a). These documents call for minimizing unnecessary risks by reducing the potential for construction debris to enter waterways.

Construction BMPs begin with proper storage of materials onsite. The materials should be stored in an area that does not freely drain to the waterbody, free from standing water or wet soil, and protected from precipitation (Lebow and Tippie 2001, NMFS 2004b). If necessary, materials should be stored on skids or support timbers to keep them off the ground. Although the wood should be BMP certified in many proposed applications, it should still be inspected on site and any pieces found to have visible residues or bleeding of preservative should be rejected. If ammoniacal treated wood has a noticeable odor, then it has not been properly processed or aged and the preservative may not be properly fixed. The wood should be rejected and the failure of the BMP certification process reported.

Maximum prefabrication should be done before the structure is placed over-water. This minimizes cutting and boring discharges of debris into the waterway. If prefabrication is done on-site, construction debris must be salvaged and disposed of properly. Cutting stations can be set up with a large tarp to capture debris. The cutting station should be kept well away from the water to minimize transport of sawdust by wind. Applications of field preservative treatments to cuts and bore holes, water repellants or other coatings, if not applied by the manufacturer at their facility, should take place at the cutting station before the wood is taken to the overwater area. These applications must be allowed to dry and/or cure.

If minimal cutting, boring or touch-up preservative applications must be performed over water, then tarps, plastic tubs or similar devices should be used to capture debris, spills or drips. Vacuums may also be used during construction to capture debris. Any excess field preservative should be wiped off and not applied in the rain. Any debris which falls into the water should be promptly removed. Debris should be stored in a dry place until it is removed from the project site. Lebow and Tippie (2001) contains useful pictures of construction BMPs.

<u>Demolition BMPs</u>

BMPs for demolition of pesticide-treated wood structures are very similar to construction BMPs. Both are meant to minimize unnecessary exposure of the aquatic environment to debris. It is recommended that minimal cutting and boring should take place over the water. Tarps, tubs and/or vacuums should be used to capture the debris. Any debris that falls into the water should be promptly removed. Additionally, wood should be stored in a dry place where the debris will not be swept away by any rising waters.

If pilings are removed, disturbance of sediments should be minimized to prevent the spread of any contamination. Sediments adjacent to the project should be analyzed to determine if they warrant removal or clean capping. Problematic sediment contamination is not necessarily from pesticide-treated wood in the infrastructure being removed, but may result from historic use at the facility (i.e. discharges from boats or industry at the site). The piles should be pulled, if possible. If pulling is not possible, the pilings should be cut at or below the sediment line and capped as warranted. Dispose of the used pilings properly with all other debris in a manner that does not expose or affect aquatic resources. Since older creosote treated wood materials were likely not produced in accordance with industry BMPs (i.e. they were likely treated to the point of refusal), they should not be reused in aquatic environments. Local requirements for disposal may vary but need to be followed.

<u>In-water Coatings and Wraps</u>
Another method to minimize unnecessary environmental risk is coating the pesticide-treated wood products with impervious materials to minimize the loss of metals or PAHs to the environment (Stillwell and Mustane 2004). Coatings or wraps should be used in projects proposed for sensitive locations, or areas with limited currents and/or high background concentrations (NMFS 2003, Hutton and Samis 2000). Examples of sensitive locations include areas with vulnerable species. These species include: Pacific herring, which may spawn onto the creosote treated wood, or pink salmon, which spawn in areas that may have maritime development (i.e. they spawn within a few miles of the coast or even within the intertidal zone). Additionally, areas determined to be important EFH for juvenile rearing should be examined carefully and coatings considered to prevent contamination especially if background levels are significantly elevated. Coatings and wraps may also be useful for piling replacement projects when a facility desires to replace a few pesticide-treated wood pilings that have been damaged, but the scale is small enough that the entire facility does not need replacement. Wraps may also be used as mitigation, to minimize impacts from existing creosote treated facilities.

There are numerous coating materials available commercially that encapsulate wood and prevent leaching of contaminants. A full review of coatings is beyond the scope of this document. The important considerations for the local biologist to consider are that the coating be inert, impervious and long lasting. The requirements are typically written for "an impact-resistant, biologically inert coating that lasts or is maintained" for a specified amount of time. Construction materials can be ordered that arrive on-site already encapsulated, or polymers may be applied by the company with the construction contract. Once dried, many coatings used in marine applications allow the piles to be driven (like an uncoated pesticide-treated wood piling). An internet search will reveal several types of products and purveyors. These types of coatings have been successfully used on projects with pesticide-treated wood pilings in San Francisco Bay since 2005 (David Woodbury, pers. comm., 2007) and in New Jersey to protect shellfish beds since that late 1990s (Stanley Gorski, pers. com., 2008). Similarly, the USACE Los Angeles District requires that creosote treated pilings used in the Ports of Los Angeles and Long Beach be wrapped to minimize the leaching of PAHs (Castanon, D. pers. Comm., 2004). Wrappings are installed and then sealed (traditionally to prevent the flux of oxygen into the wood), killing marine borers, which are already present. High density polyurethane wear strips can be used to protect the wrapped piling from damage from scraping by vessels or other objects. In California, if a project proposes to treat both in-water and above-water portions with a coating product, then it falls under the local area programmatic ESA and EFH consultation (USACE 2006) and does not require further examination by NOAA Fisheries (NMFS 2007).

Over-water Coatings

Exposed wood, used in overwater applications (such as decking) should be protected from the weather and an application of water repellent sealer is recommended by industry (WWPI 2003) and agencies (NMFS 2004b, 2003, Lebow and Tippie 2001, USDA FPL 2001). Application of finishes, such as semi-transparent penetrating stains, latex paint, or oil-based paint, decrease environmental releases (FPL 2001a and 2001b, Lebow *et al.* 2004). In general, opaque polyurethane and acrylic finishes form the most durable coatings, probably because they protect wood from ultraviolet radiation, although for some surfaces (such as those subjected to foot traffic) use of a penetrating stain that results in a slow wearing of the coating may be preferable (Stilwell and Musante 2004). Experiments on the efficacy of coatings to minimize leaching from CCA-treated wood, found that one coat of latex primer, followed by one coat of oil-based paint or two coats of penetrating, water-repellent deck stain were both effective for reducing the leaching of copper, arsenic and chromium by more than 99% (FPL 2001a). Coatings and any paint-on field treatment must be carefully applied and contained to reduce contamination (Lebow and Tippie 2001, FPL 2001b). Coatings which are likely to blister and peel or require sanding and scraping (such as varnish) should not be used for these applications. Leaching will still take place (although at a highly reduced rate) and will increase as the coating degrades. However, the rate of leaching will be greatly reduced over-time, leading to lower levels of exposure. The biologist will have to determine if the waterbody into which the contaminants are leached is sensitive enough to require that a water-proof seal or barrier must be maintained for the life of the project.

It is recommended that good construction practices, as noted elsewhere in this document, are followed with the maximum amount of construction (including coating) taking place away from any waterbodies. Lebow and Tippie (2001) of the U.S. Forest Service noted that several manufacturers of CCA and the manufacturer of another copper-treated product (known as ACQ-D) offer formulations which incorporate a water repellent into the treating solution.

Alternate Materials

Using a material other than pesticide-treated wood is another potential BMP and would eliminate the impacts of the pesticide-treated wood. Using alternative materials; however, does not eliminate the more general impacts of a structure. General structure impacts could include: shading aquatic vegetation, providing ambush cover for predatory fish and perches for piscivorous birds, introduction of pollutants from the supported vessels or industries (e.g. copper from boat hulls, PAHs from gasoline, oils and grease, turbidity from prop wash, sewage and other wastes from the vessels, industries and associated parking lots). Additional impacts from the structure may include: altering flow patterns around the vicinity of the project, changing the character of the project area (by introducing hard substrate that may be colonized by organisms not typically present) and construction impacts (such as dredging and pile driving) (NOAA 2005).

Stratus (2006a) examined several potential alternate materials. The materials have advantages and disadvantages, which are detailed in that document (Stratus 2006a). Some have their own pollutant concerns (such as leaching of potential endocrine disrupting chemicals from plastic based materials or zinc from cathode protected steel) (Xie *et al*., 2002, Weis *et al.* 1992). However, many of the dilution based arguments and assessments hold true for these materials as well. Other alternative materials may be considered nonleaching because they are made of, or coated with, nonreactive materials, but there is often a lack of data to evaluate these claims. It may be possible to use alternative materials in many of the same situations as pesticide-treated wood materials, but their prescription may not be necessary depending upon the level of impacts expected by the biologist.

Cost is often an argument brought forward in support of using pesticide-treated wood as a building material and it must be addressed here. Stratus (2006a) contains a cost comparison between pesticide-treated wood, concrete, steel and plastic pilings, which was generated by a marine construction firm under a subcontract to Stratus. The analysis concluded that concrete pilings are very cost competitive with pesticide-treated wood on an equivalent annual cost basis over the life of the project. Steel was more expensive when the analysis was conducted in 2005 and has continued to rise in price since that time. Although it is used in large projects, it may not be considered a minor change in many smaller scale projects. Plastic piling materials also had higher costs at the time of the analysis and some products may not be appropriate for some weight bearing uses. There are an expanding number of choices and capabilities in alternate materials which have improved weight bearing capacity including reinforced plastics, fusions of glass and wood and wood species which are not treated with pesticides.

Stratus (2006a) did find that concrete has a higher up-front cost (i.e. cost per pile) than pesticide-treated wood at a ratio of 2.5 to 1. However, larger scale projects that utilize concrete typically require fewer pilings. This could be attributed to the superior weight bearing properties of concrete. Some projects may be small enough that the difference in the number of pilings is inconsequential. EPA's 2008 assessment work for creosote and CCA wood preservatives includes two qualitative economic impact assessments (EPA 2008d, 2008e). However these documents do not include independent analysis of project costs using these pesticide-treated wood products compared to alternative materials. Instead, they only briefly mention one industry sponsored study (Smith 2003 in EPA 2008d, 2008e) which determined that using concrete or steel pilings was 1.96 times more expensive than using treated wood. This ratio is close to that presented by Stratus, but it can not be determined from EPA (2008d, 2008e) what factors the study considered. EPA (2008d) notes that CCA products are slightly less expensive than creosote and that creosote only accounts for 4% of the pilings market. Two case studies from the WWPI (WWPI 1998, undated) discuss two projects which also found the upfront costs of concrete piling projects to be between 2 and 2.5 times more expensive than treated wood project. These case studies do not present data examining the annual cost basis of the projects over their expected lifetimes.

The installation cost was also determined to be an important factor (Stratus 2006a). Costs may be competitive in an area with multiple pile driving companies, resulting in available equipment that can handle both pesticide-treated wood and concrete piling installations. On the other hand, areas without equipment readily available to handle the higher weight concrete pilings may see significant increases to the cost of a concrete piling project. Concrete was predicted to last longer than pesticide-treated wood (20 years compared to 15 years) by Stratus (2006a), leading to the lower cost over-time. However, these lifetime projections seem short for both products and a difference of five years may make the long term advantage less important to many smaller projects.

<u>Miscellaneous BMPS</u>
There are several other BMPs that can be beneficial in areas where the use of pesticide-treated wood products may affect a listed species or adversely impact EFH.

- Incorporate design features, which minimize abrasion of pesticide-treated wood pilings and decking. High density polyethylene wear strips can be installed on the pilings to prevent scraping by the floating docks, vessels, etc. Brooks (2004a) recommends strips that are one half-inch thick, installed down the length of the piling. As the pilings are abraded, new wood with higher leaching rates are exposed, which leads to continual unnecessary exposure to the environment. In addition to water repellent coatings (discussed earlier), decking can be protected through the use of wear guards.
- Use untreated wood for temporary structures or naturally rot resistant wood (e.g. some cedar species) for the project.
- Use top caps on creosote treated piles to minimize their exposure to the sun and subsequent losses of creosote. This should be required for all creosote treated pilings regardless of projected impact.
- Shading would also greatly reduce the amount of creosote discharged by a structure. Covering a dock will minimize the exposure of all wood products to precipitation and slow down any resultant leaching. If the project is proposed in an area with submerged aquatic vegetation (SAV), the biologist evaluating the project must take into consideration the potential impacts of shading the SAV that could result from using this BMP.
- Eliminate the use of pilings by using anchors for floating dock structures.

<u>Mitigation for Remaining Impacts</u>
Mitigation can be required, under the Magnuson-Stevens Act, for remaining impacts to EFH that can not be eliminated from a project. Mitigation may also be proposed by the project applicant as a means of reducing the uncertainty in their impact analysis, in order to facilitate the permitting process. There are a few pesticide-treated wood related mitigation options that can be considered to offset the effects of installing new pesticide-treated wood. The first is removal of old pesticide-treated wood. This can be done by removing an abandoned or unnecessary structure, or by removing pesticide-treated wood that has washed downstream as a result of flooding. NMFS (2003) contains pictures of depositional areas showing how large the old pesticide-treated wood problem can be in some locations. Another pesticide-treated wood related mitigation could involve

wrapping the already installed creosote-treated pilings in the vicinity of a project. This option is limited to creosote-treated wood. Copper-treated pilings are expected to have lost the majority of leachable copper if they have been in place for any significant period of time. Creosote-treated products, on the other hand, are expected to leach significant amounts of PAHs for years or decades, as seen in USACE (1997) and in the input parameters of the creosote leaching models. Benefits at a localized scale may be possible if a project proponent proposes to wrap pilings in the vicinity of the project, as mitigation for the unavoidable affects of the project.

Potential Exposure Scenarios

Now that the basics of pesticide-treated wood use and potential impacts have been presented, this section will present some examples of typical projects and environmental combinations that a NOAA Fisheries biologist may be asked to evaluate. Given the wide range of habitat conditions and specific life history variables that may be encountered across the Western United States, suggestions contained in this section must be coupled with site-specific information in order to make an informed decision.

Personal Use Boat Docks

A common proposed use of pesticide-treated wood is for the construction of smaller boat dock facilities. These facilities usually are constructed on waterbodies of sufficient size to provide for significant dilution potential, but it is likely that cumulative effects will need to be considered, as there are often numerous docks in these areas. Therefore, primary factors to be considered include background concentrations and stream or tidal currents. Research by Weis and Weis (1994) noted that contaminant levels in sediments associated with dock pilings, in moderately flushed areas, did not show accumulation of metals in contrast to higher surface area uses such as bulkheads. Therefore, if a personal use boat dock project can show that background levels of copper are not problematic and that densities of marine related infrastructure utilizing (or potentially utilizing) pesticide-treated wood are lower, then an individual risk assessment is not likely to be necessary. This scenario of approving smaller scale projects has already been put into place by the SLOPES III opinion (NMFS 2004b) in the lower Columbia River and the state of Oregon.

If a pesticide-treated wood bulkhead is proposed as part of the dock project, then the project may require an individual risk assessment, as recommended by industry (WWPI 2006a). In this event, there are numerous options available to the project proponent to reduce leaching. These options could include: coating the pesticide-treated wood product (just the bulkhead portion, or the whole thing), constructing the bulkhead with rock or another alternate material, or protecting the natural shoreline with vegetation or large woody debris rather than constructing the bulkhead. This could potentially eliminate the need for a site specific assessment, or in depth data gathering related to the use of pesticide-treated wood. The other potential impacts of water facilities mentioned earlier (e.g. shading of aquatic vegetation, dredging, introduction of habitats used by predatory species in the shoreline area, etc) will still need to be considered.

Marinas

Like the personal use boat dock, the construction of a marina will presumably occur on a larger waterbody. However, marinas are typically protected from currents and tidal exchange and this may result in significantly reduced dilution potential. New facilities, or facilities undergoing major renovations, are more likely to trigger some of the industry recommendations for an individual risk assessment. Examples of these projects include: installation of more than 100 pilings, construction at potentially problematic densities, or proposing a large amount of pesticide-treated wood surface area. The consulting biologist should make the project proponent aware of industry guidelines as a tool for procuring a proper effects analysis if necessary. Building or renovating marinas to "clean marina" standards is also recommended (e.g. restricting hull scraping in water, stormwater BMPs for shoreline facilities such as repair yards and parking lots, installation and mandatory use of sewage pump out stations, etc.).

Vehicle Bridges

Studies related to bridges and other overwater structures were presented earlier. The main concerns for bridges are the size and flushing rates of the waterbodies beneath the bridges, as well as the size of the bridge. The data presented earlier indicates that bridges of pesticide-treated wood are typically small enough that the footings of the bridge are not located in the water and may not even be in the 100 year floodplain. Therefore, pesticide-treated wood will only be used above the water. For a waterbody with sufficient dilution, the studies indicate that this should not be problematic and the potential impacts to salmonids are not likely to be meaningfully measured, detected or evaluated. However, extra caution may be warranted if bridge construction is proposed over a low flow stream, where significant dilution is not a given, or over a pool in a stream that supports rearing salmonids and is often disconnected during dry portions of the year. BMPs to minimize leaching (such as coatings, shading and maintenance requirements) and that minimize potential exposure, may be necessary to prevent impacting salmonids and degrading habitat.

Foot Bridges/Boardwalks

The potential impacts here are similar to the larger bridges, although some of these structures (such as long boardwalks) are more likely to use pesticide-treated wood footings that may be in the water. The main concern is for streams with periods of very limited flow, or loss of flow connectivity between pools, which lead to unacceptable exposure levels. For smaller facilities in this situation, the biologist will need to determine if coating the overwater lumber would sufficiently decrease any remaining uncertainty, or if the environment is so sensitive that an alternative decking material, such as a coated catwalk, naturally rot-resistant species of wood or plastic lumber, should be considered.

Larger facilities, such as the boardwalk examined in the Wildwood study (FPL 2000), are likely to require the generation of additional information as part of their planning process. Although the study was conducted in a sensitive environment and looked at an important potential indirect effect to salmonids (reduced prey availability), the study did not evaluate potential direct effects to salmonid olfaction. Providing technical assistance to the project proponent in earlier stages of project planning may eliminate the need for an extensive individual risk assessment in such a project.

Railroads

The vast majority of creosote treated wood (approximately 70% of all creosote use) is used in railroad construction and maintenance (EPA 2003). It is common for railroads to follow the contours of large streams and rivers in the western United States. This leads to the potential exposure of salmonid habitats from the creosote treated wood leachate. However, railroads constructed within a floodplain are typically built to minimize their chance of interacting with the water (i.e. flooding). The most likely interaction between the waterbody and railroad occurs when significant dilution is available. The most likely sources of PAHs from railroads come from normal operations (e.g. exhaust from the engines, oils and greases, herbicides used along the tracks) and from coal dust. Coal is a common cargo on many lines serving mines or coal-fired power plants.

Brooks (2004a) conducted a peer reviewed study focused on the migration of creosote from railway ties. The study showed initial leaching of creosote from new railway ties into the ballast (the rock and dirt platform upon which the tracks are laid) to approximately 60 cm in depth. This mostly occurred during the first summer after installation. There was little movement horizontally toward a constructed wetland. Following the first summer, PAH concentrations in the ballast declined (due to degradation) to background levels (at depths more than 10 cm).

PAH concentrations within the constructed wetlands' sediments were similar across treatments (new, weathered and untreated ties), indicating that atmospheric deposition of PAHs was having a large effect on the sampling results. Brooks (2004b) indicates that a small amount of PAH may have migrated into one wetland cell during the second summer of the study, as indicated by an increase in PAH of approximately 0.3 mg/kg in the sediment. In comparison, the ER-L for total PAHs is 4.0 mg/kg, making an effect from this exposure unlikely. Surface water in the wetland cells was sampled at various intervals (10 days, 2, 3, 12, and 15 months) and all samples had non-detectable concentrations until one positive sample at the 15 month stage. This sample from a new tie mesocosm contained 0.19 µg/L of benzo(a)anthracene and 0.66 µg/L of phenanthrene while a sample from an untreated tie mesocosm contained 0.16 µg/L of benzo(a)anthracene that day. These levels were not expected to be problematic as determined by the ∑TPAH methodology presented in NMFS (1998).

Brooks (2004b) does make some railroad related recommendations in order to reduce unnecessary exposure and risk to aquatic environments from railroad infrastructure. Brooks observed that numerous derelict railway crossties were discarded in the right-of-way and recommended that ties taken out of service should be disposed of properly. Due to the initial leaching observed in the study, Brooks recommended that the storage of newly treated railway ties in sensitive environments should be avoided, and the storage should occur on the ballast or on railway cars. Additionally, Brooks recommended that railway ties should be produced using management practices which minimize deep checking in the wood and excess surface deposits.

Highway Related Uses and Utility Poles

There are uses of pesticide-treated wood products in highways and roads (such as for sign posts), in addition to bridges. Utility poles are another major use of creosote-treated products, making up some 15-20% of all creosote usage (EPA 2003). These pesticide-treated wood products may be placed within the riparian area. However, like the railroads, waterways are not likely to significantly interact with these categories of pesticide-treated wood, except during flood situations. The pesticide-treated wood products may leach some contaminants when exposed to rain, or through exposure to the sun for creosote products. However, unless these products are placed over-water, or leach onto a roadway when it is raining, the leachates will likely become bound to the sediments. Contributions of copper or PAHs from these sources may not be detectable compared to the contributions coming from the road itself (e.g. oils, grease and exhaust from vehicles, copper from brake pads, spills of hazardous materials, etc.).

Conclusion

It is widely acknowledged that creosote and copper-treated wood products leach contaminants into the aquatic environment. The rate of leaching for both categories of products drops off rapidly following installation. For copper-treated products, the leaching, and resultant water column concentrations, drops off to very low levels within a few weeks to a few months, depending upon the exact product and environmental conditions. Effect level thresholds may only be exceeded for short periods of time. Copper can accumulate in sediments, where its bioavailability depends upon site-specific conditions. While the initial rate of leaching from creosote-treated pilings drops off rapidly, leaching stays elevated at easily detectable levels for many years and perhaps decades. The exact length of time this occurs is difficult to determine because the product loading and formulation of creosote utilized in the past was variable. PAHs from creosote also accumulate in sediments, where they are subject to degradation. However, the high molecular weight fraction can take a long time to degrade and contains known mutagens, teratogens, and carcinogens, which are most often associated with impacts to benthic species (e.g. tumors).

The main contaminants of concern from these products are copper and PAHs. For copper, the most sensitive sublethal endpoint may be salmonid olfaction. This may be impacted by an increase in dissolved copper concentrations as low as 0.79 μg/L above background levels. Copper may also affect salmon and EFH by reducing the quality and productivity of the benthic habitat. However, the models and studies related to copper treated wood products show the impacts are localized and only prevalent with large surface area uses (such as bulkheads) in many cases. For creosote, the main impact of concern is accumulation in the sediments. This could lead, or contribute to, elevated levels that affect the productivity of EFH, especially for groundfish species. Sediment accumulation impacts are also expected to occur on a localized scale. The impacts may occur for a longer period of time and at lower pesticide-treated wood densities than the potential impacts of copper-treated products. Water column concentrations of PAHs from creosote-treated wood are not expected to reach problematic levels, except in situations of very high density installations, or in freshwater applications. Impacts have been observed only at the most contaminated PAH sites (Eisler 2000). However, creosote-treated pilings also have the potential to impact sensitive species, which lay their eggs on the pilings (e.g. Pacific herring (Vines *et al.* 2000). Impacts could also occur in the immediate vicinity of pilings, where the PAHs accumulate in the sediments (e.g. pink salmon in Heintz *et al.* 1999). High density and/or high volume pesticide-treated wood use projects are not likely to be proposed often, due to changing patterns in pesticide-treated wood use (Brooks 2003). These types of projects are highly likely to trigger full risk assessments, and formal consultations, as recommended by industry (WWPI 2006a).

Numerous leaching studies have been conducted over the years to determine leaching rates from a variety of pesticide-treated wood formulations and for a variety of environmental conditions. The results of many of these studies were used to develop leaching models. The review of the leaching models by Stratus (2006a) found that they did an acceptable job of capturing the leaching trends and did not seem to consistently generate over or under predictions. These leaching models have been adapted into environmental prediction models, which incorporate a variety of factors (such as flow rates) to predict resultant water column concentrations and areas of sediment contamination. These models can not be relied upon to produce dependable predictions without site-specific information. However, the models seem to be useful for risk assessments, when site-specific information is available. This is due to some conservative assumptions of the models (e.g. all piles are installed simultaneously, all contaminants are considered dissolved in the water column and remain bioavailable, no dilution through turbulence of lateral dispersion, etc.(NMFS 1998))(Stratus 2006a).

The most important factor in the models' predictions is the current velocity. If significant water exchange is available to dilute the leached contaminants, then they are not predicted to increase contamination to a problematic level. Background concentrations of the contaminants are also an important consideration along with pH and project specific information (such as the number of pilings and density when installed). Especially sensitive sites are defined by species utilization (e.g. critical rearing area, entrance to a tributary), as well as environmental conditions (e.g. sediment characteristics). These sites may require special management consideration, regardless of the construction material chosen for the project.

The Poston box model, adapted by NMFS (1998), has been in use for a decade. It shows that installation of 100 or less copper-treated piles, at current velocities of 10 cm/sec or more, are not likely to result in problematic water column concentrations. However, impacts are possible with 100 piles at lower current velocities. Increases in sediment concentrations of 50% or more were not anticipated with a project of this size. 100 copper-treated pilings is also the threshold recommended by industry (WWPI 2006a) to trigger a site-specific risk assessment.

Installation of creosote-treated pilings is uncommon in the NWR and SWR of NOAA Fisheries, but is more common in the Alaska Region. Background PAH concentrations in the water column are not generally found at problematic levels with the exception of the most contaminated areas. However, NMFS (1998) predicts problematic concentrations from the installation of 100 or more creosote pilings, at most modeled installation densities, with the exception of locations with current velocities of 10 cm/sec or greater. NMFS (1998) did not present a box model for sediment contamination by PAHs because the work conducted by the USACE (1997) did not model a long-enough time span. Despite this, the effects on sediments must be considered and may affect EFH, especially for Pacific groundfish. Observed contamination at the Sooke Basin study site decreased over-time and was mostly confined to close proximity of the structures, but did result in potential effect levels in the sediments. Background concentrations need to be considered and installation of creosote in areas with elevated backgrounds should be discouraged.

Overwater uses of pesticide-treated wood products can also contribute contaminants into the aquatic environment and may be used at a high enough volume to warrant examination in a project. Copper-treated products are expected to leach most of their contamination during the first year as a result of rainfall. Creosote-treated wood will also leach in this manner, but may be expected to discharge PAHs for a longer period of time. Exposure to direct sunlight may result in the discharge of contaminants, even during the dry season, from creosote-treated products. Both categories of products may contribute additional contaminants through wear of their exposed surfaces.

BMPs are recommended as a way to reduce risk to ESA listed species and EFH. An underlying assumption in most of the leaching studies and models is that pesticide-treated wood products installed in aquatic environments will be manufactured in accordance with industry production BMPs. BMP produced wood should be used in all situations involving potential exposure to ESA listed species or EFH and is already recommended or required by several other state or Federal agencies. Conducting site-specific risk assessments, for larger projects proposing to use pesticide-treated wood, is also recommended. Industry and NMFS guidelines for copper-treated products both focus around the 100 piling size. NMFS (1998) indicates this size would be acceptable at current velocities of 10 cm/sec or greater, but can not be assumed to be protective at lower current velocities without utilizing the multiple regression equations for initial screening. Site-specific considerations on the lower Columbia River have lead to a lower threshold there (50 pilings) and many projects which typically use pesticide-treated wood (e.g. personal use boat docks) are routinely approved through this process. Potentially lower thresholds are recommended by industry in their guidance (WWPI 2006a) for some products and situations. For example, a project proposing to use 25 or more ACZA pilings parallel to the currents is recommended for a risk assessment by WWPI (2006a). The WWPI has spreadsheet based models available for use by the public (through their website) and the models were also reviewed by Stratus and found to be acceptable with the same caveats as the models used in the NMFS 1998 document.

Other BMPs, which should be routinely required, include: construction and demolition BMPs, minimization of abrasion on pilings (through the use of wear strips), use of untreated wood for temporary structures, top caps for all creosote treated pilings and proper disposal that eliminates risk to aquatic environments (while following local disposal requirements). Restricting the timing of the installation may be advisable in some locations and a simple way to eliminate potential impacts to ESA listed species. The use of coatings or wraps for pesticide-treated wood products is an acceptable method for minimizing impacts and uncertainty associated with larger-scale projects, or in especially sensitive environments. This is a necessary practice for using pesticide-treated wood products in the Southeastern U.S., because of the presence of marine borers (gribbles – *Limnoria*), which are not wholly deterred by many wood treatments. Coatings and wrappings are often used along the West Coast as well, and can even be used on the overwater portions of projects that may cause problematic levels of contamination in the aquatic environment. Exposed wood is already recommended by industry and agencies to receive an application of water repellent sealer and to be protected from the weather. The use of alternate materials can eliminate the potential impacts of pesticide-treated wood products, but can contribute some contaminants of their own. Recommending or requiring alternative materials is warranted in those situations where adverse effects can be meaningfully measured, detected or evaluated. Other BMPs may be sufficient to minimize the effects of the project making this requirement unnecessary. Mitigation options for any remaining impacts from the project should be analyzed and may be presented as EFH or ESA conservation recommendations.

Overall, the use of pesticide-treated wood products in aquatic environments with the examined formulations (ACZA, CCA and creosote) could be acceptable in many proposed projects. However, the products can not be considered categorically safe, and therefore, require project and site-specific assessment. Many projects, that still propose to use pesticide-treated wood, may pass a screening level examination and require relatively little assessment for the pesticide-treated wood related impacts. These determinations require a level of local knowledge that may be applied on a case-by-case basis, or through regional or watershed based procedures. The variability between locations makes it difficult to provide guidance on the scale of the entire west coast of the U.S. and Alaska.

The selection of copper-treated or creosote-treated products seems to be a personal preference in areas where creosote is still permitted for use. Copper-treated products are a better choice, in many instances, for minimizing impacts to NOAA Trust Resources. This is due to the rapidly diminishing level of impact and the higher sediment contamination levels needed before impacts begin to be observed. However, the limited available information shows that, in some specific instances, the proper use of creosote-treated products may not impact ESA listed salmonids in a manner that can be meaningfully measured, detected or evaluated. However, the choice of a creosote-treated product over a copper-treated product may be considered to have a greater adverse affect to the quantity or quality of EFH, especially if the product is proposed for an area which supports vulnerable species or valuable benthic habitat.

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