Appendix L

Refined Ecological Risk Evaluation

1. Introduction

On March 29, 2013, a breach in the Pegasus Pipeline operated by ExxonMobil Pipeline Company released Wabasca heavy crude oil into a residential neighborhood in Mayflower, Arkansas (Figure 1-1 of the Downstream Areas Data Assessment Report [main report]). Following this release, soil, sediment, and surface water data were collected in areas downstream of the release (downstream areas; Figures 2-1 through 2-3 of the main report). Samples were analyzed for a variety of constituents including: volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), and metals. The data from these samples were then used to conduct a screening data evaluation. Collected data were evaluated by comparing constituent concentrations to background concentrations, as well as literature-based ecological screening values (ESVs), as presented in Sections 5 through 9 of the main report. The results of this evaluation showed that a few constituents were present at concentrations above the ESVs at a small number of locations. Based on these results, a refined risk evaluation was conducted. The methods used in the refined risk evaluation are consistent with the U.S. Environmental Protection Agency (USEPA) ecological risk assessment (ERA) paradigm (USEPA 1997, 1998, 2001b). The Arkansas Department of Environmental Quality (ADEQ), which has not published state ERA guidance, refers to the USEPA guidance as a resource on their website (ADEQ 2014). This appendix provides the technical basis and results for the refined risk evaluation.

The screening data evaluation (Sections 6, 7, and 8 of the main report) involved comparing soil, sediment, and surface water analytical results to ESVs. ESVs are constituent concentrations in environmental media below which risk to ecological receptors exposed to those media is considered negligible (i.e., if concentrations do not exceed the ESV, then the constituent being screened is eliminated from any further ecological risk evaluation). However, the reverse is not necessarily true; concentrations exceeding ESVs do not automatically imply that ecological risk above acceptable levels exists, only that further ecological evaluation may be warranted (USEPA 2001a, 2001b, 2005). ESVs were identified from a variety of sources (USEPA 2001a, 2003a, 2006, 2011, 2014a, 2014b) as agreed upon with ADEQ in the Downstream Areas Remedial Sampling Plan (DARSP; ARCADIS, U.S. Inc. [ARCADIS] 2013), as described in Appendix I of main report. Further, the screening data evaluation involved calculating sample-specific toxic units (TUs) to evaluate the combined effect of PAHs in sediment and surface water on aquatic receptors (USEPA 2003b, 2008, 2012). The results of the screening data evaluation (Sections 6, 7, and 8 of the main report) found that a few constituents were at concentrations that warranted further evaluation in a number of locations, and thus the refined risk evaluation was conducted for those constituents of potential ecological concern (COPECs).

The goal of this refined ecological risk evaluation is to determine whether potential ecological risks to wildlife receptors exist, and thus guide management decisions. The refined ecological risk evaluation commences with a description of the habitat and potential ecological receptors, complete exposure pathways, and information on the sources and effects of the stressors as typical in step 1 (problem formulation) of the USEPA ERA paradigm (USEPA 1997, 1998), which is presented in Section 2. Then, based on both the problem formulation (step 1) and the results of the screening data evaluation (Sections 6, 7, and 8 of the main report), which is similar to step 2 in the USEPA ERA process, a refined risk evaluation is presented. This refined risk evaluation relies on additional lines of evidence to refine the screening data evaluation by incorporating ecological risk assessment methods typically used in step 3 of the USEPA ERA paradigm (USEPA 1997, 1998, 2000, 2001b; Figures L-1 and L-2). Specific refinements include: refinement of media and COPECs, refinement of ESVs including use of refined ESVs, refinement of exposure estimates, and risk characterization, as described in Section 3. Finally, Section 4 presents a summary of the conclusions drawn from the refined risk evaluation, which will be used to support risk management decisions for the downstream areas.

2. Problem Formulation

The problem formulation presents background information about site characterization, receptors, and ecosystem characteristics, as well as information about the sources and effects of the stressors (USEPA 1998). This information is used to develop a conceptual site exposure model that illustrates the potential relationships between stressors, pathways, and receptors such as:

- Environmental setting
- Description of media and COPECs
- Description of constituent fate and transport pathways
- Description of constituent mechanisms of ecotoxicity
- Description of potentially exposed receptors
- Identification of potentially complete exposure pathways
- Selection of generic assessment and measurement endpoints

The components of the conceptual site exposure model are described in the sections below based on the classes of compounds identified for further evaluation in the downstream areas (Sections 6 and 7 of the main report), namely VOCs and PAHs.

2.1 Environmental Setting

The ecological habitat in the downstream areas was evaluated by ARCADIS ecologists during the emergency response period and subsequently during sampling investigations. This section briefly describes the natural habitat types and qualities of those habitats in the downstream areas, and identifies wildlife that has been observed to in the downstream areas.

The downstream areas were subdivided into drainage ways, Dawson Cove, and Lake Conway. The drainage ways are a series of ditches that start at the corner of South Starlite Road and North Starlite Road and lead to a shallow drainage swale along North Main Street, which then flows east under Highway 365 and Interstate 40 (I-40) (Figures 2-1 and 2-2) into a marsh known as Dawson Cove. Dawson Cove is separated from Lake Conway by Highway 89, with water conveyed between the cove and the lake by two culverts beneath the highway (Figure 2-3).

Dawson Cove consists of a densely vegetated forested floodplain that transitioned into an open water cove south of Highway 89. The U.S. Fish and Wildlife Service (USFWS) National Wetlands Inventory (USFWS 2013a) shows Dawson Cove roughly divided into lake habitat to the north and east, and a freshwater forested/shrub wetland making up the western portion of the cove. The freshwater forest/shrub wetland is subdivided into two habitat types: forested (vegetation greater than 6 meters in height) that is temporarily flooded, and scrub/shrub (vegetation less than 6 meters in height) that is semi-permanently flooded.

Prior to vegetation removal associated with the pipeline spill, Dawson Cove was a densely vegetated forested floodplain that was dominated by an overstory of red maple (*Acer rubrum*), sweetgum (*Liquidambar styraciflua*), and deciduous trees and shrubs such as willows from the Salix genus (*Salix sp*). The understory consisted predominantly of buttonbush (*Cephalanthus occidentalis*), greenbriers (*Smilax* sp.), *Salix* sp., and poison ivy (*Toxicodendron radicans*). The shrub/scrub habitat, which is typically flooded for a longer period than the forested area, is dominated by *Salix* sp. and greenbriers. As the spill response progressed and oiled vegetation was removed from the forested portion of the cove, the cove was largely converted to an open water/open land setting. Following vegetation removal activities in the cove, Japanese millet (*Echinochloa esculenta*) was planted as ground cover.

The habitat types found in Dawson Cove extend up to the drainage ways that feed into the cove. The areas between the culverts under I-40, Highway 365, and the railroad right-of-way contain a drainage with a defined bed and bank that has surface flows for most of the year in normal-to-wet years. The vegetative species assemblage in this area is similar to the cove, with reduced overstory components and a marked increase in weedy species and habitat disturbance. The swale along North Main Street is situated in a narrow band of vegetation dominated by *Salix* sp. and has limited habitat value compared to the downstream habitat in Dawson Cove. The drainage that leads to the North Main Street swale is a concrete lined channel in a highly modified residential setting and has limited habitat value for native species.

Dawson Cove, and to a lesser extent the drainage ways, provide habitat to a variety of animal species. Reptile species (i.e., snakes, lizards, and turtles) and amphibian species (i.e., frogs, toads, and salamanders) encountered during the response are summarized in Table L-1, while Table L-2 lists some of the mammalian species encountered in Dawson Cove by Wildlife Branch staff during the response. Finally, dominant fish species found in Lake Conway are presented in Table L-3.

No federally listed threatened or endangered species are present in or adjacent to the area where the incident occurred. According to the USFWS (2013b), Faulkner County has the potential for occurrences of two listed species; piping plover (*Charadrius melodus*) and interior least tern (*Sterna antillarum athalassos*). Habitat for these two species is not present in Dawson Cove or the surrounding areas.

Arkansas is situated along the Mississippi Flyway, which is used by approximately 40 percent of North American migratory waterfowl and shorebirds during their annual migrations, as well as numerous passerine species (National Audubon Society 2013). More than 400 bird species either reside in the state year round or spend some portion of their annual migratory cycle in the state (Mills 2008). Lake Conway and Dawson Cove provide habitat to large numbers of migratory waterfowl and shore birds, as well as a large number of passerine species that make use of relatively undisturbed patches of habitat around the lake. Dawson Cove, and to some extent the drainage ways, provides habitat for a varied assortment of waterfowl, shorebirds, and passerine species. A number of waterfowl and shorebird species have been noted making use of Dawson Cove including, but not limited to, mallard (*Anas platyrhynchos*), Canadian goose (*Branta canadensis*), blue-winged teal (*Anas discors*), gadwall (*Anas strepera*), American coot (*Fulica americana*), wood duck (*Aix sponsa*), hooded merganser (*Lophodytes cucullatus*), bufflehead (*Bucephala clangula*), great blue heron (*Ardea herodias*), killdeer (*Charadrius vociferus*), Sora (*Porzana carolina*), and green-backed heron (*Butorides striatus*).

2.2 Description of Media and Constituents of Potential Ecological Concern

Soil and sediment sampling activities were conducted between July 27 and August 16, 2013, in accordance with the DARSP (ARCADIS 2013). The screening data evaluation provided in the main report (Sections 6, 7, and 8) found a few constituents at concentrations above ESVs at a small number of locations and thus, these constituents were identified as the COPECs for the refined evaluation. The COPECs, associated media, and the locations where concentrations were above ESVs are summarized below:

- Total high molecular weight (HMW) PAHs in two soil samples in the drainage ways (SO-DA-003 and SO-DA-005) and three soil samples in Dawson Cove (SO-DA-019, SO-DA-022, and SO-DA-023)

- Benzene in one soil sample in the drainage ways (SO-DA-015)
- Xylenes in three surface sediment samples (SED-DA-015, SED-DA-045, and SED-DA-048) and four subsurface sediment samples (SED-DA-015, SED-DA-017, SED-DA-039, and SED-DA-045) in Dawson Cove
- Isopropylbenzene at one subsurface sediment sample at location SED-DA-017 in Dawson Cove

Locations of the samples listed above are presented in Figures 2-1 through 2-3.

2.3 Description of Constituent Fate and Transport Pathways

Knowledge about the potential constituent fate and transport pathways is vital to understanding which constituents and receptors are associated with complete exposure pathways. This is because an ecological receptor may be exposed to a constituent in a variety of ways. In addition, the pathway and route of exposure may have a strong influence on the ecological effect of a constituent. The remainder of this subsection discusses fate and transport pathways for the classes of COPECs (i.e., VOCs and PAHs) in the downstream areas.

2.3.1 Volatile Organic Compounds: Benzene, Xylenes, and Isopropylbenzene

VOCs are often released into the environment as a result of human activities. For example, benzene and xylenes are main components of gasoline. Isopropylbenzene is found in gasoline, paints, and varnishes. Primary non-point sources of benzene, isopropylbenzene, and xylenes to the environment are aerial fallout (or rainout), road runoff (from gasoline leaks), and stormwater runoff.

VOCs, such as benzene, xylenes, and isopropylbenzene tend to attenuate rapidly in surface soil, sediment, and surface water due to their inherent volatility as their vapor pressure is high, meaning they readily volatilize. Therefore, VOCs typically have short half lives in soil, sediment, and surface water. Where volatilization is hindered (e.g., in sediment), biodegradation is expected to be an important fate process (Agency for Toxic Substances and Disease Registry [ATSDR] 2007a, 2007b). Because of these physical properties, benzene and xylenes do not bioaccumulate in the food chain (ATSDR 2007a, 2007b), and isopropylbenzene is not expected to either.

2.3.2 Polycyclic Aromatic Hydrocarbons

PAHs, much like VOCs, are released into the environment as a result of human activities such as the incomplete combustion of fossil fuels or other organic materials. Primary non-point sources of PAHs to the environment are aerial fallout (or rainout), road runoff (from the wear and leaching of asphalt, tire wear, vehicle exhaust, and dripping vehicle fluids), and stormwater runoff.

PAHs are transformed by biotic and abiotic processes in the environment. An important biotic transformation process of PAHs in soil is microbial degradation of PAHs, the rate of which is dependent on the nutrient content and bacterial community in the soil (USEPA 2007). Further, PAHs in soils undergo what is referred to as a "weathering" or "aging" process that involves volatilization of the lower molecular weight fractions and binding of the HMW fractions to organic matter in soil (USEPA 2007). Weathering can also lead to PAHs being incorporated inside the crystal lattice structure of the soil particle or soil nanopores (Ma et al. 1998). Weathering greatly affects the bioavailability of PAHs (Fairbrother 2005; Johnson et al. 2002), which in turn governs the toxicity of PAHs. In organisms, PAHs are metabolized and thus are not expected to bioaccumulate into higher trophic level wildlife (ATSDR 1995; USEPA 2007).

2.4 Description of Constituent Mechanisms of Ecotoxicity

The mechanisms of ecotoxicity for constituents vary depending on a wide range of factors, such as constituent concentrations, the receptor species exposed, the exposure route (i.e., ingestion or direct contact), and physical factors (e.g., pH, organic carbon, temperature, oxygen levels, water hardness). Some of the effects that could be observed in wildlife are mortality, reduced reproductive ability, decreased fertility, decreased offspring survival, alteration of immune and behavioral function, decreased hatching success of eggs/larvae, and retarded growth (Sample et al. 1996, USEPA 2001b). The remainder of this subsection discusses mechanisms of ecotoxicity for the COPECs. These descriptions of constituent mechanisms of toxicity are presented without consideration of constituent concentrations, as the descriptions are intended to convey an understanding of possible effects, rather than to describe the concentrations at which these effects might occur.

2.4.1 Volatile Organic Compounds: Benzene, Xylenes, and Isopropylbenzene

The toxicological effects of VOCs on ecological receptors are not well-understood, although there have been extensive inhalation studies of the effects of VOCs under laboratory conditions. Inhaled VOCs typically are metabolized in the body (often the liver), which may cause liver damage (depending on the organism) or the release of more toxic secondary metabolites. The VOCs or their metabolites may also cause neurological damage in aquatic ecological receptors (USEPA 2012), and many are mutagenic or carcinogenic. Additionally, some VOCs are fetotoxic and/or teratogenic (USEPA 2008; Sample et al. 1996).

2.4.2 Polycyclic Aromatic Hydrocarbons

Most PAHs are sorbed to solid particles in the environment, which reduces the bioavailability and toxicity of the sorbed PAHs. PAHs have been shown to cause changes in liver enzymes and to perturb cell membranes but, in general, are not viewed as acutely toxic. Some PAHs (e.g., benzo(a)pyrene) are known to be mammalian carcinogens. PAHs can have a narcotic effect on aquatic ecological receptors (USEPA 2003b). Sublethal effects attributed to PAHs in aquatic animals include reduced reproductive ability and fertility, developmental abnormalities, delayed or retarded maturation, histological changes, and carcinogenesis (Neff 1985).

2.5 Description of Potentially Exposed Receptors

The identification of receptor categories most likely to be exposed helps to focus the refined ecological risk evaluation. Potentially exposed receptors are designated based on the available habitat associated with the downstream areas, as well as the species observed there. Section 2.1 provides a description of the habitat associated with the downstream areas as well as wildlife observed in these habitats. Potentially exposed receptors include terrestrial wildlife (i.e., mammals, birds, reptiles, and invertebrates directly exposed to soil), terrestrial plants, aquatic life (fish and invertebrates directly exposed to surface water and sediment), and aquatic plants.

2.6 Identification of Potentially Complete Exposure Pathways

A complete exposure pathway is "one in which the chemical can be traced or expected to travel from the source to a receptor that can be affected by the chemicals" (USEPA 2001b). Therefore, a constituent, its release and migration from the source, a receptor, and the mechanisms of toxicity of that constituent must be demonstrated before a complete exposure pathway can be identified. The table below summarizes the potential exposure routes for the downstream areas. Wildlife receptors could be exposed to COPECs in soil and sediment though direct contact and through ingestion of food and soil/sediment particles.

Organism	Possible Exposure Routes
Terrestrial Wildlife	Ingestion, direct contact, and food chain
Terrestrial Invertebrates	Ingestion and direct contact
Terrestrial Plants	Direct contact, uptake
Terrestrial Reptiles	Ingestion, direct contact, and food chain
Aquatic Wildlife	Ingestion and direct contact, and food chain
Aquatic Life	Ingestion and direct contact
Aquatic Plants	Direct contact and uptake
Aquatic Reptiles	Ingestion, direct contact, and food chain

Amphibians and reptiles are not directly evaluated for ecological risks in this evaluation because the available toxicity data are insufficient to support establishing distinct risk-based screening criteria for such receptors, consistent with USEPA's guidance for the development of Ecological Soil Screening Levels (EcoSSLs; USEPA 2005).

Terrestrial plants and soil invertebrates are not directly evaluated for risks associated with soil COPECs because the habitat and foraging areas of wildlife that depend on them are frequently large enough to compensate for any localized losses in food or shelter, as described recently by the Texas Commission on Environmental Quality (TCEQ 2013). This logic is considered applicable to the drainage ways and Dawson Cove because of the limited number of soil COPECs (HMW PAHs and benzene), which were detected infrequently and in low concentrations within a relatively small area. Alternatively, aquatic plants typically are evaluated for exposure risks during the surface water evaluation process whereby COPEC concentrations are compared to surface water ESVs that are protective of aquatic life. For example, the surface water criteria established in the Arkansas Pollution Control and Ecology Commission (APCEC) Regulation No. 2 Standards (APCEC 2011), which are based on USEPA-derived criteria (USEPA 2014a), are protective of fish and aquatic invertebrate (e.g., daphnids) communities, as well as aquatic plants. Therefore, the screening data evaluation (Sections 6, 7, and 8 of the main report) where surface water concentrations were compared to ESVs, is considered to be inclusive of all aquatic life (i.e., fish, invertebrates, and plants).

In summary, based on the results of the screening data evaluation (Sections 6, 7, and 8 of the main report), and the discussion above regarding receptor groups, the refined risk evaluation focuses on benthic invertebrates and terrestrial wildlife. For terrestrial wildlife, the focus is on ground-level insectivores because this receptor group has the lowest soil screening levels for the COPECs and the COPECs are not bioaccumulative (USEPA 2007).

2.7 Selection of Generic Assessment and Measurement Endpoints

Assessment endpoints are the explicit expression of the ecological values to be protected (USEPA 1997). The selection of assessment endpoints depends on knowledge about the receiving environment, knowledge about the constituents released (including their toxicological properties and the relevant concentrations), and an understanding of the values that will drive risk management decisions (Suter et al. 1995). Consistent with USEPA (1998) guidance, two elements are required to define an assessment endpoint: the specific valued ecological entity and the characteristic about the entity that is important to protect. For the downstream areas, the assessment endpoints identified are the sustainability of local terrestrial and aquatic wildlife populations and communities.

USEPA Superfund guidance provides that remedial actions should be designed not to protect organisms on an individual basis, but to protect local populations and communities of biota (USEPA 1999), with the exception of threatened and endangered populations (which were not identified or expected in the downstream area, as discussed in Section 2.1). Thus, the first management principle for conducting an ecological risk assessment is to provide a basis for selecting a response action "that will result in the recovery and/or maintenance of healthy local populations/communities of ecological receptors that are or should be present at or near the site" (USEPA 1999).

Because direct measurement of assessment endpoints is often difficult or infeasible, surrogate endpoints, called measurement endpoints, are used to provide the information necessary to evaluate whether the values associated with the assessment endpoint are being protected. A measurement endpoint is defined as a measurable ecological characteristic and/or response to a stressor (USEPA 1998). Hazard quotients (HQs) and TUs typically are used as measurement endpoints in ecological risk assessments. HQs are defined as the ratio of the concentration of a given constituent to its ESV. TUs are analogous to HQs. Toxicity of nonionic organic compounds (e.g., PAHs) in sediment to benthic organisms can be estimated by using a TU, which is the ratio of the predicted concentration in porewater to an ecotoxicity-based effects concentration. Ratios for individual PAHs are summed to determine a TU for the PAH mixture. As such, HQs and TUs greater than one indicate that there is potential for ecological risk while HQs and TUs below one are an indication that risk to ecological receptors is not expected. While HQs were not calculated in the screening data evaluation (Sections 5, 6, and 7 of the main report), the evaluation utilized the same concept when concentrations were compared to ESVs. The sample-specific TUs calculated in the screening data evaluation (Sections 6, 7, and 8 of the main report) can be considered measurement endpoints. The refined risk evaluation follows the methods used in the screening evaluation where exposure estimates (concentrations) are compared to ESVs.

3. Refinement of Ecological Risk

The refined ecological risk evaluation is designed to identify the nature and extent of ecological risks in order to support informed risk management decision-making. This approach refines the conservative screening data evaluation (Sections 6, 7, and 8 of the main report), which was designed to rule out further evaluation of constituents and media that clearly do not pose a significant ecological risk. The refined ecological risk evaluation provided herein is analogous to step 3a of a baseline ecological risk assessment (USEPA 1997, 1998). Step 3a focused only on the COPECs and media that progress beyond the screening step. Because further ecological evaluation was indicated, it was conducted with the intent to be an incremental iteration of exposure, effects, and risk characterization, which is in accordance with USEPA's ERA guidance (USEPA 2001b). Therefore, the assumptions used in refined risk evaluation are refinements of the conservative estimates of exposure and ecotoxicity screening values (USEPA 2001b). This culminated in the refined risk evaluation using a lines-of-evidence approach in accordance with USEPA guidance (1997, 1998, 2001b). The refined risk evaluation involved the following steps:

- Identification of media of concern/habitat
- Refinement of COPECs
- Refinement of ESVs
- Refinement of exposure estimates
- Refinement of risk evaluation

A discussion of each step is provided in the following subsections (Sections 3.1 through 3.4). Then all of the refinements are considered for the COPECs identified in the screening data evaluation to provide a refined risk evaluation (Section 3.5).

3.1 Identification of Media of Concern/Habitat

The media of concern and habitat that are the focus of this refined ecological risk evaluation have been identified. The media of concern (soil and sediment) and their limited spatial extent were identified in the screening data evaluation (Sections 6 and 7 of the main report) and the type and quality of habitat is identified in Section 2.1, above.

3.2 Refinement of Constituents of Potential Ecological Concern

The process for refining the COPECs can involve various steps (USEPA 2001b), including:

1. Comparison with background and upgradient concentrations.

2. Frequency of detection – Constituents detected in greater than 5 percent of the samples in a given medium typically are retained as COPECs and considered in the next step of the refinement process.

The evaluation of data for the drainage ways and Dawson Cove compared to background data was presented in the screening data evaluation (Sections 6, 7, and 8 of the main report). The locations and constituents that were identified for further evaluation in the screening data evaluation are summarized in Section 2.1. The evaluation of frequency of detection will be discussed in the risk characterization step of this refined ecological risk evaluation (Section 3.5).

3.3 Refinement of Exposure Estimates

The exposure estimate is a representative concentration of a constituent in an environmental medium that is potentially contacted by the ecological receptor. In the screening data evaluation (Sections 6, 7, and 8 of the main report), the medium-specific constituent concentrations in each sample were used as the exposure estimate. This approach is appropriate for sessile receptors (e.g., infaunal sediment invertebrates), which have limited to no ability to move across an area for foraging or other activities. For mobile receptors (e.g., mammals, birds), using the medium-specific constituent concentration as the exposure estimate is considered conservative, and it is appropriate to establish refined exposure estimates for these receptors that are protective of local ecological populations. As recommended by USEPA's ERA guidance (1997 and 1998), a mean concentration is the preferred exposure estimate for mobile receptors. While USEPA's ERA guidance does not specify an appropriate measure of the mean, USEPA human health risk guidance recommends utilizing the 95 percent upper confidence limit on the mean (95% UCL) as the exposure concentration (USEPA 1989, 2002). The UCL is a statistical number calculated to represent the mean concentration with a high level (e.g., 95 percent) of confidence that the true arithmetic mean concentration will be less than the UCL. The high level of confidence is used to compensate for the uncertainty involved in representing site conditions with a finite number of samples.

As stated above, USEPA ERA guidance does not specify how the appropriate measure of the mean should be estimated. However, recently released guidance by the TCEQ entitled “Determining Representative Concentrations of Chemicals of Concern for Ecological Receptors” (TCEQ 2013) elaborates on the methods recommended for estimating the exposure concentration for ecological receptors. TCEQ guidance contends that the exposure estimates should be based on the exposure area and ecological habitat. The guidance further elaborates that the exposure area should not be defined by a receptor’s home range (TCEQ 2013), which is in agreement with USEPA’s ERA guidance that calls for the protection of wildlife populations/communities and also the measurement endpoints selected herein for the refined risk evaluation. Therefore, in this refined risk evaluation, the exposure area for terrestrial wildlife receptors was assumed to be the terrestrial habitat in the downstream areas (i.e., soil in the drainage ways and soil in Dawson Cove).

Refined soil exposure estimates for wildlife receptors were calculated for use in the refined risk evaluation. The sample by sample exposure estimate approach used in the screening data evaluation is an overly conservative approach for wildlife that move around because they will not be exposed to a single location 100 percent of the time. Thus, a refined exposure estimate based on the 95% UCL is a more reasonable exposure estimate for wildlife.

For the identification of exposure estimates, the 95% UCL was calculated and used as the refined exposure concentration. The 95% UCLs were calculated using the most recent version of USEPA’s statistical software ProUCL (Version 5.0; USEPA 2013). To calculate the exposure estimate, only surface soil samples (0 to 0.5 foot below ground surface [bgs]) were utilized. While that defines the biologically active zone in sediment, the biologically active zone in soil is often deeper to account for borrowing animals. However, as a conservative measure (to avoid underestimating the exposure concentration) data from the 0 to 0.5 foot depth interval in soil was used to represent the exposure estimate in soil (surface and subsurface) for terrestrial wildlife because the source of the COPECs is a surface spill rather than a subsurface release, and most of the locations with concentrations above ESVs were in surface soil. The refined exposure estimates were used in the risk characterization (Section 3.5).

3.4 Refinement of Ecological Screening Values

ESVs are generally based on effects such as mortality and reproductive impairment, and are assumed to be widely applicable to sites throughout the United States for screening purposes (USEPA 1997). For most constituents and receptors, the data available to generate ESVs are limited and related to effects on individual organisms, rather than subpopulations or communities. Given these limitations, conservative assumptions are used to ensure that the ESVs are protective. Where ESVs are published, users are cautioned to recognize that such screening values do not constitute remediation goals, as they are sometimes based on highly conservative exposure assumptions and/or wildlife receptors that may not be applicable to a particular site (USEPA 2001a, 2005). As such, their robustness and biological association with the assessment endpoint may be limited.

In the refinement of ESVs, an expanded range of ESVs is utilized, where available and appropriate, to provide insight into the type or likelihood of impacts. Refined ESVs utilized in the refined risk evaluation were obtained from sources listed in the hierarchy established for identifying ESVs (Section 5 of the main report and Appendix I) that was established in the DARSP (ARCADIS 2013) based on ADEQ recommendations. Refined ESVs used in this refined risk evaluation are discussed in the following subsections and will be employed in the refined risk characterization (Section 3.5).

3.4.1 Refined Ecological Screening Values for Soil

The only COPECs identified in soil were benzene and HMW PAHs. Refined soil ESVs for both benzene and HMW PAHs identified for use in this refined ecological risk evaluation are discussed below.

Benzene

The ESV for benzene in soil of 10 micrograms per kilogram (µg/kg) utilized in the screening data evaluation (Section 6 of the main report) was identified from the draft USEPA Region 4 ecological screening values (USEPA 2011). This value (10 µg/kg) is a proposed value (USEPA 2011) that updates the prior value of 50 µg/kg in the previous version of the Region 4 screening values table (USEPA 2001a). Both values (i.e., 10 µg/kg and 50 µg/kg) are not USEPA-derived values, but are Target Values developed by the Dutch Ministry of Housing, Spatial Planning, and Environment (MHSPE) in the document entitled Circular on Target Values and Intervention Values for Soil Remediation (MHSPE 2000) for use in screening soil to ensure protection of human health and the environment. The Dutch MHSPE identifies both Target Values and Intervention Values; the Target Value for benzene is 10 µg/kg, while the Intervention Value is 1,000 µg/kg. As noted in the MHSPE (2000) document, these values are based on an integration of the human and ecotoxicological effects for each chemical. MHSPE (2000) does not provide a set of values that are protective of human health and another set of values that are protective of ecological receptors. The Target Values are meant to be protective of both human health and the environment, and are intended to represent a relatively 'unpolluted' soil (MHSPE 2000). As such, the Dutch MHSPE Target Values are not considered applicable for situations where the receptors of interest are limited to terrestrial wildlife.

A refined ESV for benzene of 255 µg/kg was identified from USEPA Region 5 (USEPA 2003), which is based on modeled exposure and risk to a masked shrew (*Sorex cinerus*) through incidental soil ingestion and uptake through diet. Other refined ESVs considered were screening levels from the American Petroleum Institute (API) that were calculated for a variety of livestock including dairy cattle, beef cattle, calves, sheep, goats, camels, and horses (API 2004). API livestock ESVs for benzene ranged from 756 milligrams per kilogram (mg/kg; horse) to 2,198 mg/kg (calves). However, because the receptors in question in the downstream area are smaller mammals, the ESV from Region 5 was used as the refined ESV in the refined ecological risk evaluation.

<u>HMW PAHs</u>

The ESV for HMW PAHs in soil was identified from the USEPA EcoSSL guidance document for PAHs (USEPA 2007), as described in Appendix I. EcoSSLs are calculated by combining dietary uptake models and conservative toxicity thresholds (i.e., toxicity reference values [TRVs]) that rely on generic receptor exposure assumptions that may not reflect site receptors or conditions. USEPA EcoSSLs are highly conservative and only intended as a screening tool.

The ESV of 1.1 mg/kg for HMW PAHs in soil is based on exposure to mammals, specifically for a ground insectivore, as represented by a shrew, as that was the lowest (and therefore the most protective) mammalian EcoSSL (USEPA 2007). In contrast, the EcoSSL for the herbivore receptor group (represented by a vole) was 39 mg/kg and the EcoSSL for the carnivore receptor group (represented by a weasel) was 100 mg/kg. The EcoSSL for the shrew was calculated using a conservative TRV, and conservative uptake and bioavailability assumptions. The TRV used by USEPA to calculate the HMW PAH EcoSSL is based on a no observed adverse effect level in mice exposed to benzo(a)pyrene equal to 0.615 milligrams per kilogram body weight per day (mg/kg bw/day) selected from the USEPA database for PAH toxicity (USEPA 2007). Likewise, the bioaccumulation factor (2.6) used to derive the HMW PAH EcoSSL was selected to be conservative and is two orders of magnitude higher than what is generally seen in the field (Jager et al. 2003). Where high bioaccumulation factors have been used to estimate risk to herbivorous and insectivorous mammals, adverse effects were not documented in the field when concentrations exceeded the EcoSSLs (Kapustka 2004a). Additionally, bioaccumulation factors two orders of magnitude lower than those used to derive the EcoSSL have been observed in the field (Krauss et al. 2000; Ma et al. 1998). Moreover, the EcoSSLs currently available from USEPA (2014b) do not account for bioavailability, although bioavailability plays an important role in uptake and toxicity (Interstate Technology & Regulatory Council [ITRC] 2011).

The discussion provided above illustrates that the ESV for HMW PAHs based on the EcoSSL is very conservative. A refined ESV for HMW PAHs could be calculated using a geometric mean TRV, other TRVs based on PAHs delivered with food (Kapustka 2004b), site-specific bioaccumulation estimates, or other refinements. However, for the purposes of this refined risk evaluation, the EcoSSL will be conservatively used as the refined ESV.

3.4.2 Refined Ecological Screening Values for Sediment

Constituents with concentrations that were above ESVs in sediment (COPECs) were isopropylbenzene and xylenes. In the refined risk evaluation, refined sample-specific ESVs for those COPECs were used.

The screening ESVs for xylenes and isopropylbenzene were obtained from the USEPA Region 3 screening values (USEPA 2006), consistent with the hierarchy described in Appendix I. These ESVs were based on the equilibrium partitioning sediment benchmarks (ESBs; USEPA 2012), which in turn were calculated based on water quality benchmarks (WQBs) from USEPA Region 3 and assumed default total organic carbon content in sediment. These ESBs were based on the standard approach for calculating ESBs for nonionic organic compounds (USEPA 2008), which uses the mass fraction of organic carbon in sediment (f_{OC}), the constituent-specific partition coefficient between water and organic carbon (K_{OC}), and the WQB as shown in the equation below:

$$Standard\ ESB = WQB \times K_{OC} \times f_{OC}$$

The standard approach assumes that the bioavailable fraction of non-polar organic constituents is equivalent to the fraction of the sediment concentration that is freely dissolved in interstitial water, and that the freely dissolved fraction is determined primarily by the extent of partitioning to organic carbon. This equation applies only to sediments having ≥ 0.2 percent total organic carbon by dry weight and nonionic organic chemicals with logarithm of the octanol-water partition coefficient (K_{OW}) ≥ 2 (USEPA 2008).

However, the standard ESB equation (provided above) is ineffective for assessing less-hydrophobic organic compounds in sediment (e.g., xylenes and isopropylbenzene) because it fails to account for the contribution of dissolved constituent to the total constituent concentration in sediment. For constituents having low K_{OC} values, this results in sediment quality benchmarks that are more conservative than benchmarks calculated by assuming 100 percent bioavailability.

For less hydrophobic compounds (that is, compounds having log octanol-water partition coefficient [K_{OW}] values below 3.5), Fuchsman (2003) developed an equation to estimate a modified ESB (MESB) that is recommended by USEPA in the *Compendium of Tier 2 Values for Nonionic Organic Compounds* (USEPA 2008).

$$MESB = WQB \times \left[(K_{OC} \times f_{OC}) + \left(\frac{1 - f_{solids}}{f_{solids}}\right)\right]$$

Where:

MESB = Modified equilibrium partitioning sediment benchmark (micrograms per kilogram);
WQB = Water quality benchmark (micrograms per liter);
f_{OC} = Fraction of sediment present as organic carbon (unitless);
K_{OC} = Organic-carbon/water partition coefficient (liter per kg); and
f_{solids} = Fraction of sediment present as solids (unitless).

Sample-specific ESVs (i.e., MESBs) for xylenes and isopropylbenzene were calculated using sample-specific measured values for f_{OC} and f_{solids} and refined WQBs using the approach recommended by USEPA (2008). For example, an refined WQB was identified for xylenes because the Region 3 WQB used to calculate the Region 3 sediment benchmark is a value developed more than 17 years ago by Suter and Tsao (1996), and was based on a very small toxicity dataset for xylenes that was limited to test results from only two species (i.e., *Pimephales promelas* [fathead minnow] and *Lepomis macrochirus* [bluegill]). In the intervening years, additional aquatic toxicity data for xylenes have become available. Therefore, the WQBs developed by the TCEQ for the protection of aquatic life were used as refined WQBs (TCEQ 2006). The TCEQ WQBs for xylenes and isopropylbenzene are protective of freshwater aquatic life via chronic exposure conditions, and were derived using aquatic toxicity data for species indigenous to Texas, whenever possible, and also from the most sensitive species. For example, the aquatic toxicity dataset used by TCEQ for xylenes included results for *Pimephales promelas* (fathead minnow), *Lepomis macrochirus* (bluegill), *Cyprinus carpio* (carp); *Poecilia reticulata* (guppy), *Carassius auratus* (gold fish), and *Oncorhynchus mykiss* (rainbow trout). These species, along with their associated aquatic toxicity data and the TCEQ-derived WQBs, are considered appropriate for exposure conditions encountered in Mayflower, Arkansas.

3.5 Refinement of Risk Evaluation

The refined risk evaluation utilized several lines of evidence to evaluate risk in the downstream areas. Using several lines of evidence to characterize risk provides both a process and a framework for reaching a conclusion regarding confidence in the risk estimate (USEPA 1998). The lines of evidence used include consideration of: frequency of detection, fate and transport characteristics, use of refined exposure estimates, use of refined ESVs, evaluation of spatial extent, and finally considerations of habitat quality.

3.5.1 Soil in the Drainage Ways

The screening data evaluation (Section 6 of the main report) found benzene in one soil sample in the drainage ways (SO-DA-015) at concentrations above the ESV, and soil PAH concentrations, specifically HMW PAHs, greater than ESVs at two soil sampling locations in the drainage ways (SO-DA-003 and SO-DA-005). The refined risk evaluation for these COPECs is discussed in the paragraphs below.

A total of 45 soil samples were collected in the drainage ways. Benzene was detected in 7 percent of those samples (3 of 45) and was above the conservative ESV of 10 µg/kg in only one of those samples (SO-DA-015 at 0 to 0.5 foot bgs). However, that detection was below the refined ESV of 255 µg/kg identified from USEPA Region 5 (USEPA 2003a; see Section 3.4.1). Further, the sample was collected from a vegetated strip of land between I-40 and Highway 365 (Figure 2-2), while higher quality terrestrial habitat is available nearby around Dawson Cove and therefore, will be preferentially used by ecological receptors. In conclusion, risk to ecological receptor populations from exposure to benzene in soil in the drainage ways is not expected due to the low frequency of detection and because the detected concentration is below the refined ESV.

To refine the risk from potential exposure to HMW PAHs in soil in the drainage ways, the PAH exposure estimate (i.e., the 95% UCL; see Section 3.3) was compared to the screening ESVs (EcoSSL). The 95% UCL, calculated based on 14 surface soil (0 to 0.5 foot bgs) samples, was 1.02 mg/kg (Table L-4), which is lower than the EcoSSL of 1.1 mg/kg. Further, the two samples where concentrations individually were above the EcoSSL (SO-DA-003 and SO-DA-005) were collected in the swale along North Main Street (Figure 2-1), which has limited habitat value and is subject to runoff sources from the roadway, whereas high quality habitat is available nearby and therefore, therefore be preferentially used by ecological receptors. In conclusion, risk to ecological receptor populations from exposure to PAHs in soil in the drainage ways is not expected.

3.5.2 Soil in Dawson Cove

The screening data evaluation (Section 6 of the main report) found soil HMW PAH concentrations greater than ESVs in three soil samples in Dawson Cove (SO-DA-019, SO-DA-022, and SO-DA-023). As part of the refined evaluation, the Dawson Cove soil HMW PAH exposure estimate was calculated and compared to the ESV. The 95% UCL, based on 15 surface soil samples, was 0.78 mg/kg (Table L-5), which is below the conservative screening ESV based on the EcoSSL of 1.1 mg/kg. This indicates that risk to ecological receptor populations from exposure to PAHs in soil in Dawson Cove is not expected.

3.5.3 Sediment in Dawson Cove

The screening data evaluation (Section 7 of the main report) for Dawson Cove sediments found three surface samples (SED-DA-015, SED-DA-045, and SED-DA-048) and four subsurface samples at locations SED-DA-015, SED-DA-017, SED-DA-039, and SED-DA-045 with concentrations of xylenes greater than the ESV of 25.2 mg/kg. Xylene concentrations were either non-detect or below the sediment ESV in the remaining samples (118 of 125 samples). Further, the results of the screening data evaluation indicated that one subsurface sample at location SED-DA-017 also contained isopropylbenzene at a concentration greater than the ESV of 86 mg/kg in Dawson Cove. Isopropylbenzene concentrations either were non-detect or below the sediment ESV in 124 of 125 samples.

In the refined risk evaluation, these sediment concentrations were compared to refined sample-specific ESVs calculated based on sample-specific organic carbon content (Section 3.4.2), as presented in Table L-6. Concentrations of isopropylbenzene in both the surface and subsurface samples at location SED-DA-017 were found to be below the sample-specific ESVs. Similarly, concentrations of xylenes were also below the sample-specific ESVs in all samples. Additionally, the xylene levels have most likely been reduced since the sampling took place due to natural processes such as volatilization and degradation. Because concentrations of xylenes and isopropylbenzene were below refined ESVs, they had a low frequency of detection, and they tend to attenuate rapidly in the environment, the risk to benthic invertebrate communities from exposure to xylenes and isopropylbenzene in Dawson Cove sediment is not expected.

4. Summary and Conclusions

Risks were characterized for terrestrial and aquatic receptors at the drainage ways area and at Dawson Cove based on a refined evaluation that included several lines of evidence including refined exposure area concentrations (for soil), refined and refined media-specific ESVs, low frequency of detection for the COPECs, and limited spatial extent of COPEC occurrence. The conclusions drawn based on the refined risk evaluation are provided below:

- Benzene and HMW PAHs were identified as soil COPECs. However, based on several lines of evidence, no adverse effects are expected for mammalian populations from exposure to the COPECs in soil at the drainage ways area and Dawson Cove.

- Xylenes and isopropylbenzene were identified as COPECs for sediment. However, based on the lines of evidence evaluated (e.g., use of refined ESVs, and low frequency of detection), adverse effects are not expected for the aquatic life community in Dawson Cove sediment.

Based on these results and in accordance with USEPA guidance (1997), there is adequate information to conclude that there is no unacceptable risk to ecological receptor populations from the COPECs and no further ecological assessment for the drainage ways area and Dawson Cove is warranted.

5. References

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Tables

Table L-1
Amphibian and Reptile Species Encountered in Dawson Cove

Downstream Areas Data Assessment Report
ExxonMobil Environmental Services Company
Mayflower Pipeline Incident Response, Mayflower, Arkansas

Common Name	***Scientific Name***
Amphibians	
Frogs and Toads	
American bull frog	*Lithobates catesbeianus*
Blanchard's cricket frog	*Acris blanchardi*
Coastal plains leopard frog	*Lithobates sphenocephala utricularius*
Dwarf American toad	*Anaxyrus americanus charlesmithi*
Eastern narrow mouth toad	*Gastrophryne carolinensis*
Fowler's toad	*Anaxyrus fowleri*
Green frog	*Lithobates clamitans*
Green tree frog	*Hyla cinerea*
Salamanders	
Marbled salamander	*Ambystoma opacum*
Mole salamander	*Ambystoma talpoideum*
Spotted salamander	*Ambystoma maculatum*
Three-toed amphiuma	*Amphiuma tridactylum*
Western lesser siren	*Siren intermedia nettingi*
Reptiles	
Snakes	
Broad-banded watersnake	*Nerodia fasciata confluens*
Eastern garrter snake	*Thamnophis sirtalis sirtalis*
Eastern hog-nose snake	*Heterodon platirhinos*
Midland brown snake	*Storeria dekayi wrightorum*
North American racer	*Coluber constrictor ssp.*
Northern rough green snake	*Opheodrys aestivus aestivus*
Orange striped ribbon snake	*Thamnophis proximus proximus*
Plain-bellied watersnake	*Nerodia erythrogaster*
Prairie kingsnake	*Lampropeltis calligaster calligaster*
Rat snake	*Pantherophis sp.*
Rattlesnake	*Crotalus sp.*
Southern copperhead	*Agkistrodon contortrix contortix*
Speckled king snake	*Lampropeltis holbrooki,*
Western cottonmouth	*Agkistrodon piscivorus leucostoma*
Western mud snake	*Farancia abacura reinwardtii*
Western rat snake	*Pantherophis obsoletus*
Western worm snake	*Carphophis vermis*
Lizards	
Broadhead skink	*Plestiodon laticeps*
Common five lined skink	*Plestiodon fasciatus*
Prairie lizard	*Sceloporus consobrinus*
Southern coal skink	*Plestiodon anthracinus pluvialis*
Turtles	
Common snapping turtle	*Chelydra serpentina*
Eastern musk turtle	*Sternotherus odoratus*
Eastern river cooter	*Pseudemys concinna concinna*
Mississippi map turtle	*Graptemys pseudogeographica kohni*
Ornate box turtle	*Terrapene ornata ornata*
Red-eared slider	*Trachemys scripta elegans*
Southern painted turtle	*Chrysemys dorsalis*
Spiny soft shell turtle	*Apalone spinifera ssp.*
Three-toed box turtle	*Terrapene carolina triunguis*
Western chicken turtle	*Deirochelys reticularia miaria*

Table L-2
Mammalina Species Encountered and/or Expected in Dawson Cove

Downstream Areas Data Assessment Report
ExxonMobil Environmental Services Company
Mayflower Pipeline Incident Response, Mayflower, Arkansas

Common Name	*Scientific Name*
Beaver	*Castor canadensis*
Field mouse	*Mus sp.*
Grey squirrel	*Sciurus carolinensis*
Mink	*Neovison vison*
Muskrat	*Ondata zibethicus*
Nine-banded armadillo	*Dasypus novemcintus*
Nutria	*Myocastor coypus*
Raccoon	*Procyon lotor*
Short tailed shrew	*Blarina brevicauda*
Striped Skunk	*Mephitis mephitis*
Virginia opossum	*Didelphis virginiana*
White-tailed deer	*Odocoileus virginianus*
Woodchuck	*Marmota monax*

Table L-3
Dominant Game and Forage Fish Species Found in Lake Conway and Dawson Cove

Downstream Areas Data Assessment Report
ExxonMobil Environmental Services Company
Mayflower Pipeline Incident Response, Mayflower, Arkansas

Common Name	*Scientific Name*
Black bullhead	*Ameriurus melas*
Black crappie	*Pomoxis nigromaculatus*
Blue catfish	*Ictalurus furcatus*
Bluegill	*Lepomis macrochirus*
Bowfin	*Amia calva*
Brown bullhead	*Ameriurus nebulosus*
Chain pickerel	*Esox niger*
Channel catfish	*Ictalurus punctalus*
Common carp	*Cyprinus carpio*
Flathead catfish	*Pylodictis olivaris*
Freshwater drum	*Aplodinotus grunniens*
Grass carp	*Ctenopharyngodon idella*
Green sunfish	*Lepomis cyanellus*
Hybrid bream	(cross between bluegill and green sunfish)
Largemouth bass	*Micropterus salmoides*
Longeared sunfish	*Lepomis megalotis*
longnose gar	*Lepisosteus osseus*
Redeared sunfish	*Lepomis microlophus*
Shad	*Alosa sp.*
Smallmouth buffalo	*Ictiobus bubalus*
Warmouth	*Lepomis gulosus*
White bass	*Morone chrysops*
Yellow bullhead	*Ictalurus natalis*

Table L-4
ProUCL Output for High-Molecular Weight PAHs in Drainage Way Soil

Downstream Areas Data Assessment Report
ExxonMobil Environmental Services Company
Mayflower Pipeline Incident Response, Mayflower, Arkansas

General UCL Statistics for Data Sets with Non-Detects

User Selected Options	
From File	For ProUCL_a.wst
Full Precision	OFF
Confidence Coefficient	95%
Number of Bootstrap Operations	2000

Result (total hmw pahs drainage way)

General Statistics

Number of Valid Observations	14	Number of Distinct Observations	14
Raw Statistics		**Log-transformed Statistics**	
Minimum	48.3	Minimum of Log Data	3.877
Maximum	2280	Maximum of Log Data	7.732
Mean	579.5	Mean of log Data	5.768
Geometric Mean	320	SD of log Data	1.185
Median	276		
SD	656.7		
Std. Error of Mean	175.5		
Coefficient of Variation	1.133		
Skewness	1.746		

Relevant UCL Statistics

Normal Distribution Test		**Lognormal Distribution Test**	
Shapiro Wilk Test Statistic	0.779	Shapiro Wilk Test Statistic	0.967
Shapiro Wilk Critical Value	0.874	Shapiro Wilk Critical Value	0.874
Data not Normal at 5% Significance Level		Data appear Lognormal at 5% Significance Level	
Assuming Normal Distribution		**Assuming Lognormal Distribution**	
95% Student's-t UCL	890.3	95% H-UCL	1793
95% UCLs (Adjusted for Skewness)		95% Chebyshev (MVUE) UCL	1511
95% Adjusted-CLT UCL (Chen-1995)	955.7	97.5% Chebyshev (MVUE) UCL	1906
95% Modified-t UCL (Johnson-1978)	904	99% Chebyshev (MVUE) UCL	2682
Gamma Distribution Test		**Data Distribution**	
k star (bias corrected)	0.814	Data appear Gamma Distributed at 5% Significance Level	
Theta Star	712.2		
MLE of Mean	579.5		
MLE of Standard Deviation	642.4		
nu star	22.78		
Approximate Chi Square Value (.05)	12.93	**Nonparametric Statistics**	
Adjusted Level of Significance	0.0312	95% CLT UCL	868.2
Adjusted Chi Square Value	11.95	95% Jackknife UCL	890.3
		95% Standard Bootstrap UCL	851.1
Anderson-Darling Test Statistic	0.321	95% Bootstrap-t UCL	1218
Anderson-Darling 5% Critical Value	0.761	95% Hall's Bootstrap UCL	2095
Kolmogorov-Smirnov Test Statistic	0.179	95% Percentile Bootstrap UCL	886.1
Kolmogorov-Smirnov 5% Critical Value	0.235	95% BCA Bootstrap UCL	956.1
Data appear Gamma Distributed at 5% Significance Level		95% Chebyshev(Mean, Sd) UCL	1345
		97.5% Chebyshev(Mean, Sd) UCL	1676
Assuming Gamma Distribution		99% Chebyshev(Mean, Sd) UCL	2326
95% Approximate Gamma UCL (Use when n >= 40)	1021		
95% Adjusted Gamma UCL (Use when n < 40)	1105		
Potential UCL to Use		Use 95% Approximate Gamma UCL	1021

Note: Suggestions regarding the selection of a 95% UCL are provided to help the user to select the most appropriate 95% UCL.
These recommendations are based upon the results of the simulation studies summarized in Singh, Singh, and Iaci (2002)
and Singh and Singh (2003). For additional insight, the user may want to consult a statistician.

Table L-5
ProUCL Output for High-Molecular Weight PAHs in Dawson Cove Soil

Downstream Areas Data Assessment Report
ExxonMobil Environmental Services Company
Mayflower Pipeline Incident Response, Mayflower, Arkansas

General UCL Statistics for Data Sets with Non-Detects

User Selected Options

From File	For ProUCL.wst
Full Precision	OFF
Confidence Coefficient	95%
Number of Bootstrap Operations	2000

Result (total hmw pahs dawson cove)

General Statistics

Number of Valid Observations	15	Number of Distinct Observations	15

Raw Statistics		Log-transformed Statistics	
Minimum	5.59	Minimum of Log Data	1.721
Maximum	1760	Maximum of Log Data	7.473
Mean	547.8	Mean of log Data	5.493
Geometric Mean	243.1	SD of log Data	1.804
Median	361		
SD	511.3		
Std. Error of Mean	132		
Coefficient of Variation	0.933		
Skewness	1.119		

Relevant UCL Statistics

Normal Distribution Test		Lognormal Distribution Test	
Shapiro Wilk Test Statistic	0.884	Shapiro Wilk Test Statistic	0.816
Shapiro Wilk Critical Value	0.881	Shapiro Wilk Critical Value	0.881
Data appear Normal at 5% Significance Level		Data not Lognormal at 5% Significance Level	
Assuming Normal Distribution		**Assuming Lognormal Distribution**	
95% Student's-t UCL	780.4	95% H-UCL	9294
95% UCLs (Adjusted for Skewness)		95% Chebyshev (MVUE) UCL	3289
95% Adjusted-CLT UCL (Chen-1995)	805.8	97.5% Chebyshev (MVUE) UCL	4286
95% Modified-t UCL (Johnson-1978)	786.7	99% Chebyshev (MVUE) UCL	6244
Gamma Distribution Test		**Data Distribution**	
k star (bias corrected)	0.635	Data appear Normal at 5% Significance Level	
Theta Star	862.7		
MLE of Mean	547.8		
MLE of Standard Deviation	687.5		
nu star	19.05		
Approximate Chi Square Value (.05)	10.15	**Nonparametric Statistics**	
Adjusted Level of Significance	0.0324	95% CLT UCL	765
Adjusted Chi Square Value	9.363	95% Jackknife UCL	780.4
		95% Standard Bootstrap UCL	757.5
Anderson-Darling Test Statistic	0.59	95% Bootstrap-t UCL	839.2
Anderson-Darling 5% Critical Value	0.776	95% Hall's Bootstrap UCL	830
Kolmogorov-Smirnov Test Statistic	0.222	95% Percentile Bootstrap UCL	767.7
Kolmogorov-Smirnov 5% Critical Value	0.23	95% BCA Bootstrap UCL	790
Data appear Gamma Distributed at 5% Significance Level		95% Chebyshev(Mean, Sd) UCL	1123
		97.5% Chebyshev(Mean, Sd) UCL	1372
Assuming Gamma Distribution		99% Chebyshev(Mean, Sd) UCL	1861
95% Approximate Gamma UCL (Use when n >= 40)	1028		
95% Adjusted Gamma UCL (Use when n < 40)	1115		
Potential UCL to Use		Use 95% Student's-t UCL	780.4

Note: Suggestions regarding the selection of a 95% UCL are provided to help the user to select the most appropriate 95% UCL.
These recommendations are based upon the results of the simulation studies summarized in Singh, Singh, and Iaci (2002)
and Singh and Singh (2003). For additional insight, the user may want to consult a statistician.

Table L-6
Refined Evaluation of Constituents of Potential Ecological Concern Sediment

Downstream Areas Data Assessment Report
ExxonMobil Environmental Services Company
Mayflower Pipeline Incident Response, Mayflower, Arkansas

Analyte	Sediment Sample									
	SED-DA-015 0-0.5 ft (µg/kg)	SED-DA-015 0.5-1 ft (µg/kg)	SED-DA-017 0-0.5 ft (µg/kg)	SED-DA-017 0.5-1 ft (µg/kg)	SED-DA-039 0-0.5 ft (µg/kg)	SED-DA-039 0.5-1 ft (µg/kg)	SED-DA-045 0-0.5 ft (µg/kg)	SED-DA-045 0.5-1 ft (µg/kg)	SED-DA-048 0-0.5 ft (µg/kg)	SED-DA-048 0.5-1 ft (µg/kg)
Physical Parameter										
Percent Moisture	29.5	26.4	23.6	25.0	27.9	24.9	53.2	40.7	59.3	53.0
Total Organic Carbon (TOC)	2.15	NA	1.88	NA	1.93	NA	4.64	NA	6.41	NA
f_{oc}	0.0215	0.0215	0.0188	0.0188	0.0193	0.0193	0.0464	0.0464	0.0641	0.0641
f_{solids}	0.7050	0.7360	0.7640	0.7500	0.7210	0.7510	0.4680	0.5930	0.4070	0.4700
Refinement of Comparison to ESVs[1]										
Xylene (Total)										
Concentration (µg/kg)	730	420	8	2,600	2 J	3 J	110	61	81 J	5 J
Sample Specific ESV (µg/kg)	11,595	11,515	10,062	10,095	10,424	10,349	25,337	24,733	34,850	34,408
Isopropylbenzene										
Concentration (µg/kg)	--	--	1 J	280 J	--	--	--	--	--	--
Sample Specific ESV (µg/kg)	--	--	3,425	3,431	--	--	--	--	--	--

Notes:

1. The sample concentration was compared to a sample-specific ESV calculated based on the equilibrium partitioning approach (USEPA 2008):
   Sample-specific ESV (µg/kg) = WQB (µg/L) x [(f_{oc} x K_{oc})+ (1-f_{solids}/f_{solids})].
   Where:
   WQB = water quality benchmark (µg/L) from TCEQ (2006) of 1,340 and 255 µg/L for xylene and isopropylbenzene, respectively.
   f_{oc} = fraction organic carbon (TOC presented in decimal form).
   K_{oc} = organic carbon partition coefficient (unitless) from ORNL (2013); 383 and 698 for xylene and isopropylbenzene, respectively.
   f_{solids} = fraction solids (1 - moisture content in decimal form).
   TOC for the subsurface samples (0.5 to 1 ft) was set at the measured value in the surface samples (0 to 0.5 ft) at same location.

COPEC = constituent of potential ecological concern
ESV = ecological screening value
ft = foot/feet
TOC = total organic carbon
µg/kg = microgram per kilogram
µg/L = microgram per liter

References:

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U.S. Environmental Protection Agency (USEPA). 2008. Procedures for the Derivation of Equilibrium-Partitioning Sediment Guidelines (ESGs) for the Protection of Benthic Organisms: Compendium of Tier 2 Values for Nonionic Organics. United States Environmental Protection Agency, Office of Science and Technology, Office of Research and Development. EPA/600/R-02/016. March.