TRANSCONTINENTAL GAS PIPE LINE COMPANY, LLC

NORTHEAST SUPPLY ENHANCEMENT PROJECT

COMPLIANCE STATEMENT

APPENDIX L. EVALUATION OF RISKS TO ECOLOGICAL RECEPTORS DUE TO RE-SUSPENDED CONTAMINANTS

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Evaluation of Risks to Ecological Receptors due to Resuspended Contaminants
Northeast Supply Enhancement Project
October 2017

Introduction

Transcontinental Gas Pipe Line Company, LLC (Transco) proposes to construct the Northeast Supply Enhancement Project (Project), which includes the installation of pipeline and associated subsea facilities along a 23.33-mile offshore route that crosses Raritan Bay, Lower New York Bay, and the Atlantic Ocean in New Jersey and New York. Release of sediments during the pipeline installation process has the potential to impact nearby water-borne and benthic biota through physical and chemical effects.

During pipeline installation, sediments suspended by dredging and jetting activities could result in physical effects on organisms in the water column through the clogging of gills or abrasion. Physical impacts could also include smothering when the suspended sediments are redeposited adjacent to the disturbed areas. Physical effects are discussed in greater detail in other Project documents, such as Resource Report (RR) 3 of Transco's application to the Federal Energy Regulatory Commission (FERC) for a Certificate of Public Convenience and Necessity (Certificate), as well as Transco's draft Essential Fish Habitat (EFH) Assessment, which was submitted as an appendix to RR3. The August 2017 report titled *Hydrodynamic and Sediment Transport Modeling Results – Base-Case Simulations* (sediment modeling report) and an October 2017 addendum to the report describe the potential distribution of suspended sediments and associated deposition due to Project-related dredging/jetting activities.

Based on samples collected by Transco in late 2016, some portions of the offshore route traverse sediments with elevated levels of contaminants. The entrainment of these sediments during dredging/jetting activities will result in the dispersion and redeposition of these sediments into adjacent areas. The fate of these contaminated sediments may result in chemical effects on both biota in the water column and benthic organisms. The following discussion presents Transco's analysis of the potential for ecological impacts from exposure of biological receptors to contaminants released during dredging/jetting activities. See Transco's June 2017 *Offshore Environmental Sampling Report*, submitted to FERC on June 30, 2017, and subsequent addendum and errata for a complete description of Transco's Project-specific sampling effort and the results of associated sediment chemistry analyses.

Sediment Contaminant Concentrations

Sediment chemical analyses indicate that, in general, elevated contaminants are concentrated at a limited number of discrete locations. More specifically, the data from vibracores collected during the late 2016 survey and analyzed at 3-foot intervals indicate that most of the sample sites have at least one contaminant concentration that exceeded either New York and/or New Jersey 'upper-level effects' sediment screening thresholds, identified as 'Class C' and 'Effects Range – Medium' (ERM), respectively. However, the elevated contaminant levels were typically restricted to the upper 3 feet of the seabed. Concentrations of organic contaminants were greater than 'upper-level effects' thresholds at approximately a third of the sample sites, with at least one inorganic (metal) threshold exceeded at approximately 83 percent of the sample sites.

Results for representative heavy metals (i.e., cadmium, copper, lead, zinc, and mercury) known to have deleterious effects on aquatic biota such as fish (Authman et al. 2015) indicate that one sample location had upper-threshold exceedances for mercury only; one sample location had upper-threshold exceedances for lead and mercury; two sample locations had upper-threshold exceedances for lead, zinc, and mercury; and one sample location had upper-threshold exceedances for copper, lead, and mercury. There were no exceedances of the upper-threshold for cadmium.

It is important to note that these metals occur naturally in sediment, and are generally associated with the fine-grained fraction of sediment (EPA 2007). Based on an aluminum-normalization approach, the U.S. Environmental Protection Agency (EPA) calculated that in 1993 and 1994 approximately 82% of the surface sediment in the Lower New York Bay, Raritan Bay and Sandy Hook Bay complex was anthropogenically enriched in at least one metal (i.e., zinc) (Adams et al. 1998). On the other hand, the EPA estimated that only approximately 36% of the surface sediment in this waterbody complex was anthropogenically enriched in cadmium.

Of the 69 sample sites along the proposed offshore route (see Figure 1), Sites VC-1, VC-16, and VC-17 exhibited a greater number of exceedances of established thresholds for several contaminants at all depth intervals and, therefore, have been considered as having the worst-case sediment conditions along the route. The exceedances at these sample sites were for dioxins, polychlorinated biphenyls (PCBs), select polycyclic aromatic hydrocarbons (PAHs), and metals (including mercury). Although upper-level thresholds were exceeded for several of these contaminants, greatest concentrations were never more than one order-of-magnitude (i.e. ten times) above a threshold concentration.

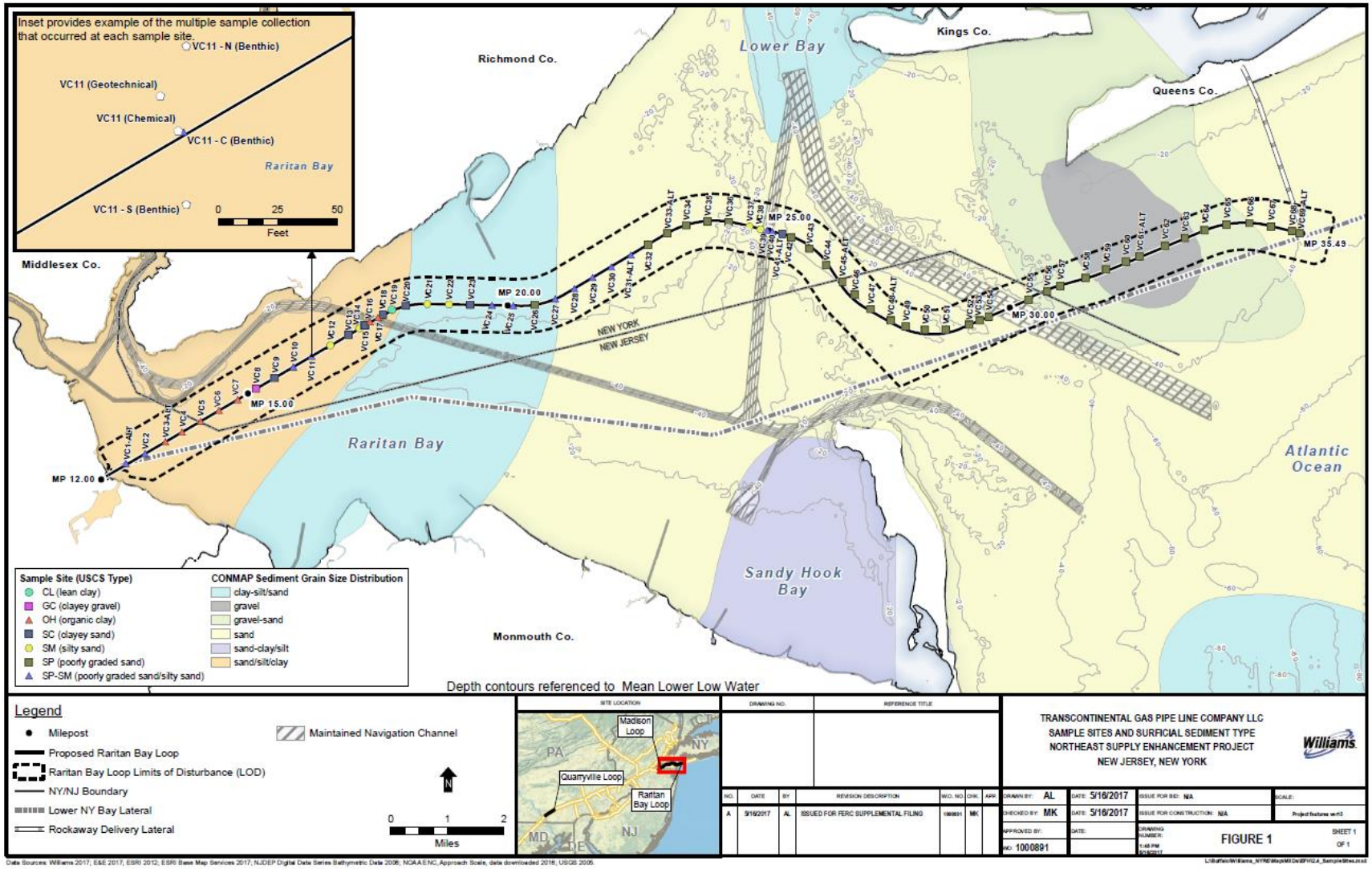
Table 1 presents the maximum concentrations of (total) mercury and total PCB congeners that were detected in sediments from these three sample sites; the potential biological effects from the resuspension of these two representative contaminants are discussed further below. Note that the volume of sediment released into the water column during Project-related dredging at these particular locations is expected to be 5% or less of the total disturbed material.

Table 1
Comparison of Select Sediment Contaminant Concentrations from Select Sample Sites to New York and New Jersey 'Upper-level Effects' Sediment Screening Thresholds.

Contaminant	NYSDEC Class C (mg/kg)	NJDEP Medium Effects Range (mg/kg)	Maximum (mg/kg)	Mean (mg/kg)
			VC-1-ALT (all depths)	
Total PCB Congeners	1.0	0.18	1.60	0.39 (N=10)
Mercury	0.71	0.71	5.28	1.4 (N=11)
			VC-16 (all depths)	
Total PCB Congeners	1.0	0.18	2.0	0.796 (N=5)
Mercury	0.71	0.71	4.22	2.33 (N=5)
			VC-17 (all depths)	
Total PCB Congeners	1.0	0.18	0.73	0.356 (N=6)
Mercury	0.71	0.71	3.77	1.83 (N=6)

Key:
kg = kilogram
mg = milligrams
N = number of samples
NJDEP = New Jersey Department of Environmental Protection
NYSDEC = New York State Department of Environmental Conservation
PCB = polychlorinated biphenyls

Figure 1: Sample Sites and Surface Sediment Type



Contaminant Fate Background

The expected fate and subsequent potential of a contaminant to cause biological effects is based on the physicochemical conditions found within the sediment matrix and the contaminant's molecular structure and stability. For example, within the sediment matrix, physicochemical factors affecting bioavailability include grain size, redox potential (Eh), pH, and the presence of organic carbon, sulfides, and other compounds (Hamelink et al. 1992; Eggleton and Thomas 2004). Adsorption of both metals and organics to organic carbon can limit bioavailability (Haitzer et al. 1998), but metals can be released from the sediment matrix when exposed to reduced pH conditions and oxygenated waters. Therefore, the total concentration of a contaminant in sediment is not a reliable predictor of a chemical's potential impact on an ecosystem (Gourlay-Francé and Tusseau-Vuillemin 2013). Again, metal concentrations in sediments may be indicative of their naturally occurring relationship to grain size or organic carbon and may not necessarily represent toxic levels (EPA 2007).

During dredging/jetting, physical and chemical changes in the sediment occur simultaneously, affecting the chemical forms and bioavailability of contaminants such as metals and organic compounds. The generally accepted assumption is that only the dissolved fraction of a chemical is bioavailable to organisms that concentrate pollutants from water through direct contact and respiration (Landrum et al. 1985). A chemical's bioavailability is based on its chemical structure, which in turn dictates its behavior within the various environmental compartments (i.e., sediment, porewater, surface water). For metals, this chemical structure is based on the speciation of the metal. Exchangeable (dissolved) cations can have high mobility within environmental compartments if ion exchange occurs. Metals from anoxic sediments will be resuspended, resulting in variable desorption rates of metals adsorbed to sulfides (e.g., mercury, lead, copper are released more rapidly than zinc) (Caille et al. 2003). Metals co-precipitated with or adsorbed to iron sulfide and manganese sulfide are rapidly oxidized (within the first few minutes) following sediment resuspension due to their relative solubility in oxygenated conditions (Allen et al. 1993; Simpson et al. 1998; Caetano et al. 2002).

Generally, hydrophobic organic contaminants readily desorb from sediment, although the rate of desorption tends to decrease with time (Chen et al. 2000; Lamoureux and Brownawell 1999; Zhang et al. 2000). Desorption rates and times also depend on the size of the sediment particles and, in the case of PCBs, the degree of chlorination (Borglin et al. 1996). Highly chlorinated PCBs or hydrophobic contaminants with large partition coefficients tend to desorb more slowly, with desorption occurring over several years (Gourlay-Francé and Tusseau-Vuillemin 2013).

Thermodynamic characteristics affect a molecule's tendency to move between environmental compartments based on relative concentrations (i.e., fugacity) (Clark and McFarland 1991); chemicals shift towards equilibrium and thus diffuse from areas of higher fugacity to those of lower fugacity. This is termed 'partitioning', which is represented by molecule-specific mathematical constants (i.e., coefficients) that reflect the concentration differential between two compartments (Clark and McFarland 1991). The primary partition coefficients that are used to describe the behavior of organic chemicals in the environment include: octanol:water (K_{ow}); organic carbon:water (K_{oc}); aqueous solubility (S), and bioconcentration factor (K_B). These properties are important for assessing a chemical's potential for bioaccumulation. Some chemicals are broken down and easily eliminated by organism and do not bioaccumulate (e.g., PAHs in fish), while others are readily taken up and cannot be broken down and eliminated (PCBs) (Clark and McFarland 1991).

Project Risk Assessment Approach

Based on an understanding of the expected fate mechanisms for metals and organics released from suspended sediments, it is possible to predict the risk to benthic communities and other aquatic organisms from contaminant exposure. For this Project, two approaches have been taken to predict contaminant-related risk. First, a qualitative evaluation was conducted that predicts how relevant the risk may be to local marine/estuarine biota from exposure to metals after resuspension during dredging/jetting activities. While this qualitative evaluation considers mercury in particular, it generally addresses metal-specific traits that are important for understanding the fate of metals and, ultimately, the risk they could pose. Mercury was selected based on the likelihood that its occurrence near the Project results from anthropogenic sources (i.e., is not naturally occurring) (Eisler 1987), and the higher likelihood that upper-level mercury threshold exceedances may result in impacts on aquatic organisms compared to other metals (Adams et al. 1998). Indeed, the EPA estimated that approximately 71% of Lower New York Bay, Raritan Bay and Sandy Hook Bay contained mercury levels that were determined to be anthropogenically enriched (Adams et al. 1998). Secondly, for neutral organics (e.g., PCBs) a discussion of expected fate is provided along with a screening-level model used to evaluate the potential for biouptake of PCBs by two benthic taxa that are important components of the New York Bight ecosystem: the hard clam, *Mercenaria*, and a ubiquitous polycheate, *Nereis virens*.

<u>Mercury (and other metals)</u>

In natural environments, mercury (Hg) can exist in three oxidation states: elemental Hg, Hg(I), and Hg(II); the most toxic form, methyl mercury, occurs when elemental mercury becomes methylated via microbial action within the sediment (Eisler 1987). To predict the potential ecological risk from the distribution of mercury (and other metals) from contaminated sediment during dredging/jetting activities,

the likely chemical state of mercury prior to and after sediment disturbance has been considered. In sediments, mercury tends to associate with organic matter, and under aerobic conditions mercury may combine with sulfur to form insoluble sulfides (Jaagumagi 1990). Although the standard EPA laboratory analytical method (SW-846/7474) for sediment measures total mercury, and most sediment screening values are based on total mercury concentrations, Kannan et al. (1998) and Taylor et al. (2012) respectively found that methyl mercury accounted for only 0.77% and 0.42% of the total mercury found in sediments in Florida and in Narragansett Bay. Methylated mercury poses a much greater risk to biological organisms than elemental mercury (Eisler 1987), but as noted above, it can be expected to comprise less than 1% of the total reported mercury in sediment. These studies suggest that the vast majority of mercury in sediments at the Project site will likely be the less-toxic elemental form.

Mercury (along with other metals) will quickly disassociate from sulfide complexes into the water column after dredging (Benoit et al. 1999). In laboratory studies, Stern and Stickle (1978) found that a majority (97.4%) of the mercury released will be adsorbed on clay and organic particles within the water column; their field studies determined that biologically available mercury levels within the water column were negligible. Although Burks and Engle (1978) found that heavy metals can become soluble during dredging operations, LaSalle et al. (1991) determined that reduced iron from sediments will actively scavenge these metals (and other compounds) after release, causing them to settle to the bottom where they are again reduced under anoxic conditions. Studies by Forstner et al. (1989) and Reible et al. (2002) found that in situations where Eh and pH do not change dramatically, the release of metals during dredging is negligible. Because of the buffering capacity in the energetic offshore marine environment near the Project, conditions will likely favor consistent Eh and pH throughout all disturbance activities. Chapman et al.'s (1998) study of a harbor at the U.S. submarine base in Scotland found that although sediments were highly contaminated with PCBs and heavy metals (copper and zinc), bioassays indicated no acute toxicity to water column taxa from the release of bound contaminants. Similarly, the U.S. Department of Transportation (2004) noted that entrainment and deposition of metals (and organics) during dredging operations within New York Harbor would not pose a risk to benthic macroinvertebrates or fish. All of these physicochemical mechanisms, along with documented studies, suggest that the entrainment and redistribution of sediments contaminated with mercury (and other metals) during the Project's dredging/jetting activities will not pose a risk to biota in the water column. Furthermore, even in areas where depositional amounts may affect benthic macroinvertebrates from a physical perspective, the tendency of entrained metals to associate with particulate sediments, organic carbon, and ferrous oxides within an oxygenated environment will limit their bioavailability within the depositional sediment matrix. Van den Berg et al. (2001) and De Groote et al. (1998) found low mobilization of metal contaminants into the dissolved phase during dredging. This was generally thought to be due to the rapid

scavenging of sulfide-liberated metals by newly formed iron and manganese oxides/hydroxides. Their studies were consistent with Bonnet et al.'s (2000) simulated dredging study where no or low levels of metal contaminants were released and concentrations returned to background levels within 27 hours.

The sediment modeling indicates that high-energy conditions in the offshore Project area may redistribute contaminated sediment up to several hundred feet from the sediment-disturbed areas. However, such conditions also result in lower contaminant concentrations due to dilution across a larger geographic extent (Rice and White 1987). Furthermore, similar levels of contamination are expected to be present in the existing surface sediments at areas in and near the offshore workspaces. For example, approximately 29% of the Lower New York Bay, Raritan Bay, and Sandy Hook Bay sites sampled by the EPA in 1993 and 1994 contained sediments with mercury concentrations that exceeded the ERM threshold, while 32% of the sites sampled in these bays exceeded the ERM threshold in 1998 (Adams and Benyl 2003). Based on local conditions along the offshore Project route, including the relatively limited distribution of 'upper-level' exceedances for mercury and other heavy metals along the Project route, the expected fate of metals within the water column after release and redeposition, the dilution and redistribution of sediments, and the high likelihood for metals to be quickly bound after redeposition (see LaSalle et al. 1991), the risk to offshore ecological communities from mercury (and other metals) exposure is predicted to be low.

Organics

The fate of organic compounds in sediment after dredging will vary based on molecular structure, size, and resulting partitioning coefficients. The sediment matrix will play a significant role in predicting whether an organic compound becomes mobile following disturbance. Hydrophobic organic contaminants generally desorb from sediments over time. PCB congeners tend to desorb at different rates, with a small fraction thought to be desorption-resistant (Lamoureux and Brownawell 1999). Typically, organics do not mobilize far from the source and result only in localized sediment concentration increases (Rice and White 1987). As noted above, high-energy conditions in the offshore Project area may mobilize sediment and increase the extent of contaminant redistribution, but these conditions also result in lower contaminant concentrations due to dilution across a larger geographic extent (Rice and White 1987).

Although marine species exposure to released organics is limited based on particle absorption and dilution factors, Transco has opted to look at worst-case bioaccumulation by benthic organisms that could be exposed to PCBs near the Project using the Theoretical Bioaccumulation Potential (TBP) model (McFarland and Clarke 1999). TBP has commonly been used as the first-step in evaluating dredged sediment for open-water disposal (McFarland and Clarke 1999). Per McFarland and Clark (1999), "Prior

to laboratory bioaccumulation testing, TBP is usually performed. The TBP model estimates the probable concentration of a neutral organic chemical that would eventually bioaccumulate in an organism from continuous exposure to the sediment. The assumption of the TBP model derives from thermodynamics: the system consisting of sediment, organism and water is visualized as being closed. Total organic carbon (TOC) in the sediment and lipid in the organism are assumed to be the primary compartments that account for the partitioning of neutral organic compounds. Thus, the concentration of a chemical in the sediment is normalized on the basis of sediment organic carbon content, and the application of a partition coefficient enables calculation of the expected equilibrium concentration in an exposed organism of stated lipid content (McFarland 1984; McFarland and Clarke 1986)." The TBP model equation is:

$$TBP = BSAF\ (C_s / fOC)\ fL$$

where: BSAF is the biota/sediment accumulation factor;
C_s is the concentration of neutral organic chemical in sediment (in mg/kg);
fOC is the decimal fraction total organic carbon in sediment; and
fL is the decimal fraction lipid content of the target organisms.

The TBP model was applied to provide a conservative bioaccumulation estimate of total PCBs from sediment to the hard clam (*M. mercenaria*) and to sandworms (*N. virens*). PCBs were the selected organic compounds based on their high potential for biouptake and biotransfer within marine food chains. Tables 2 and 3 present, respectively, the TBP modeling results for *M. mercenaria* and *N. virens* using the maximum and mean total PCB concentrations found in sediments at Site VC-16, which is where the highest PCB concentrations were detected along the offshore route. The PCB BSAFs were obtained from the Biota-Sediment Accumulation Factor Database, which is maintained by the Environmental Laboratory of the U.S. Army Engineer Research and Development Center (https://bsaf.el.erdc.dren.mil/index.cfm). Lipid data was obtained from McFarland et al. (1994).

Table 2
TBP model input parameters and results for PCBs biouptake in *M. mercenaria* from maximum and mean sediment concentrations (mg/kg)

Sample Site	Contaminant	Sediment Concentration (C_s, mg/kg)	BSAF	fOC (decimal TOC)	fL (decimal lipid)	TBP Concentration (mg/kg)
VC-16 (9-12 feet)	Total PCBs	2.00[1]	0.245[2]	0.032[3]	0.008	0.123
VC-16 (Average)	Total PCBs	0.796[4]	0.245[2]	0.032[3]	0.008	0.049

[1] Maximum from VC-16, 9-12 ft. interval.
[2] From MacFarland et al. 1994; represents *M. mercenaria* BSAF for total PCBs.
[3] From VC-16, average TOC value from all (5) sample intervals.
[4] Mean from VC-16, all intervals.

Table 3
TBP model input parameters and results for PCBs biouptake in *N. virens* from maximum and mean sediment concentrations (mg/kg)

Sample Site	Contaminant	Sediment Concentration (C_s, mg/kg)	BSAF	fOC (decimal TOC)	fL (decimal lipid)	TBP Concentration (mg/kg)
VC-16 (9-12 feet)	Total PCBs	2.00[1]	0.0344[2]	0.032[3]	0.042	0.090
VC-16 (Average)	Total PCBs	0.796[4]	0.0344[2]	0.032[3]	0.042	0.036

[1] From VC-16, 9-12 ft. interval.
[2] From MacFarland et al. 1994; represents polychaete BSAF for total PCBs.
[3] From VC-16, average TOC value from all (5) sample intervals.
[4] Mean from VC-16, all intervals.

The results of the TBP model suggest that hard clam PCB whole-body concentrations based on maximum and mean sediment concentrations at VC-16 would likely be low compared to highly contaminated sites such as New Bedford Harbor, where viable populations of hard clam are present, but the lowest tissue concentrations were much greater and modeled uptake could exceed 200 mg/kg (Judd et al. 2014). Recently, the Massachusetts Department of Environmental Protection (MDEP) found an average concentration of 1.3 milligrams per kilogram (mg/kg) for all PCB congeners in hard clams measured in New Bedford Harbor Area 1 (within the harbor) (MDEP 2015). Similarly, Rubinstein et al. (1983) conducted PCB accumulation studies on benthic species using four sediment types from New York Harbor. For sediment PCB concentrations ranging from 0.46 to 7.28 mg/kg, hard clam tissue never exceeded 1.3 mg/kg during the 100-day study period. After 100 days, all body burdens fell to 0.6 mg/kg or less for all sediments tested, and variability between sediment types was low (Rubinstein et al. 1983). In comparison, the New Jersey Department of Environmental Protection (NJDEP 2001) found PCB concentrations in hard clams in Raritan Bay ranging from 0.0138 to 0.0392 mg/kg (wet weight). Thus, considering average PCB concentrations at the most contaminated location (VC-16), TBP-derived concentrations in hard clam tissue from Project-related sediment redeposition could be expected to be similar to or less than historical tissue concentrations (per NJDEP 2001) after completion of the Project. Importantly, the model prediction is very conservative as the majority of sediments across the Project site have PCB concentrations much lower than those found at VC-16.

Rubinstein et al. (1983) also assessed uptake of PCBs by the sandworm, *N. virens*. His results indicated greater variability in uptake with different sediment types and resultant maximum tissue concentrations of 0.6 (mean of 0.4) mg/kg after 60 days of exposure. Project-derived concentrations for the sandworm using the highest and mean PCB concentrations at VC-16 were an order-of-magnitude less than those noted by Rubinstein et al. (see Table 3). These findings suggests that the redistribution

of PCBs found in Project sediments will not pose a risk to upper-tier organisms within the marine food web of the Project Area.

Conclusions

Based on this analysis, Transco predicts that there is a low risk of adverse effects on ecological receptors from exposure to metals and organic contaminants in sediment that will be suspended in the water column and redeposited during Project-related dredging/jetting activities. Throughout the offshore Project area, overall surficial and at-depth concentrations of both metals and organic compounds are relatively low compared to contaminated sediments in harbors across the U.S. More specifically, comparison to both New York and New Jersey sediment screening criteria resulted in moderate exceedances of upper-level thresholds, and exceedances were never more than one order-of-magnitude greater than either state's thresholds. Also, fate characteristics for released metals and organics during dredging/jetting activities are expected to result in only short-term water quality changes near the point of disturbance, as most contaminants will be adsorbed to organic material and fine-grained sediment, and redeposited as bound compounds. Furthermore, sediment with higher contaminant levels will be mixed with adjacent less-contaminated material and dispersed away from the point of sediment disturbance, resulting in dilution of the contaminants. The diluted contaminant levels in the redeposited material are expected to be similar to ambient contaminant concentrations in surface sediments at the depositional locations.

The results of TBP modeling using maximum PCB concentrations measured along the offshore route in 2016 suggest that the entrainment and redeposition of even the most contaminated sediments along the route will not adversely affect local biota or food webs. Clarke and McFarland (2000) noted that TBP was a useful screening tool for eliminating sediments having a negligible likelihood of causing unacceptable effects because of bioaccumulation from further testing, and that the model tended to generally overestimate hydrophobic organic contaminants' bioaccumulation from sediment. Considering the conservative nature of the TBP model and the expected fate of compounds currently present in the sediment along the route, Transco expects the redistribution of sediments disturbed by during the Project will not result in effects on local biota. Additionally, the 2016 sediment sampling results indicate that the redistributed sediments will be generally consistent with the composition of surface sediments near the dredged/jetted sites.

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