LOTT Clean Water Alliance
Reclaimed Water Infiltration Study

ECOLOGICAL RISK ASSESSMENT

FINAL

Prepared for

LOTT Clean Water Alliance
500 Adams Street NE
Olympia, WA

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Acronyms

ACR	acute-to-chronic ratio
AWQC	ambient water quality criteria
BAF	bioaccumulation factor
bw	body weight
cfs	cubic feet per second
COPEC	chemical of potential ecological concern
CSM	conceptual site model
DBCP	dibromochloropropane
dw	dry weight
EAE	ecological assessment endpoint
EC50	concentration that causes a nonlethal effect in 50% of an exposed population
Eco-SSL	ecological soil screening level
ED20	effect dose associated with a 20% reduction of growth, reproduction, or survival
EDB	ethylene dibromide
EPA	US Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
FAV	final acute value
FCV	final chronic value
GMAV	genus mean acute value
HQ	hazard quotient
LC50	dose that is lethal to 50% of an exposed population
LOAEL	lowest-observed-adverse-effect level
LOEC	lowest-observed-effect concentration
LOTT	LOTT Clean Water Alliance
MATC	maximum allowable toxicant concentration
MIA	molecular initiating event

MOA	mode of action
MWRWP	Martin Way Reclaimed Water Plant
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
O3-BAC-GAC	ozone-biologically activated carbon filtration-granular activated carbon
PBDE	polybrominated diephenyl ether
PFAS	per- and polyfluoralkyl substances
PFBA	perfluoro butanoic acid
PFBS	perfluoro-1-butanesulfonic acid
PFD	problem formulation document
PFHxA	perfluoro-n-hexanoic acid
PFNA	perfluoro-n-nonanoic acid
PFOA	perfluoro octanoic acid
PFPeA	perfluoropentanoic acid
RO-AOP	reverse osmosis-advanced oxidation processes
ROC	receptor of concern
RWIS	reclaimed water infiltration study
SMAV	species mean acute value
SSD	species sensitivity distribution
TCPP	tris(chloropropyl)phosphate
TDCPP	tris(1,3-dichloro-2-propyl)phosphate
TRV	toxicity reference value
UCL	upper confidence limit
USFWS	US Fish and Wildlife Service
Windward	Windward Environmental LLC

Executive Summary

This document presents the ecological risk assessment (ERA) for the LOTT Clean Water Alliance (LOTT) reclaimed water infiltration study (RWIS). The purpose of the RWIS is to evaluate the use of reclaimed water for groundwater recharge. The ERA assesses the potential risk posed by residual chemicals (e.g., pharmaceuticals and chemicals found in household and personal care products) to aquatic-dependent organisms that utilize nearby streams where groundwater discharges.

The ERA was conducted in accordance with US Environmental Protection Agency (EPA) guidance. Chemicals of potential ecological concern (COPECs) were initially identified through a screening-level evaluation. The list of COPECs was refined using data from a subsequent groundwater fate and transport model, and a final list of five COPECs was evaluated in detail in the exposure analysis and effects and risk characterizations. The ERA found that LOTT's proposed use of reclaimed water for groundwater recharge does not pose unacceptable risk to aquatic-dependent organisms that utilize nearby streams where groundwater discharges.

ES.1 Problem Formulation

The first phase of the ERA, the problem formulation, was conducted in 2019–2020 (Windward 2020). The problem formulation included a site description for the two waterbodies of interest (Woodland and McAllister Creeks), selection of receptors of concern (ROCs), development of a conceptual site model (CSM), identification of assessment and measurement endpoints, and identification of COPECs.

The primary study area for the RWIS is composed of the Hawks Prairie Ponds and Recharge Basins in northeast Lacey, Washington. At the study site, Class A reclaimed water produced by LOTT is conveyed through a series of constructed wetland ponds into recharge basins, where it is infiltrated through the soil and into groundwater. From the Hawks Prairie site, groundwater in the shallow aquifer flows to the southwest toward Woodland Creek. A portion of groundwater migrates from the shallow aquifer to the sea-level aquifer and flows toward McAllister Creek.

ROCs for Woodland and McAllister Creeks include the general aquatic community that may be exposed to residual chemicals via direct contact with surface water (e.g., aquatic plants, invertebrates, fish, and herptiles), as well as fish and aquatic-dependent wildlife that may feed in Woodland and McAllister Creeks. Belted kingfisher and northern river otter were selected as ROCs to represent piscivorous species of birds and mammals, respectively.

The CSM describes pathways through which ecological receptors may be exposed to residual chemicals and identifies assessment endpoints and risk questions to evaluate those endpoints. The most significant pathways evaluated in the ERA are direct exposure to surface water, exposure of fish from bioaccumulation of chemicals in tissue, and exposure through ingestion of fish tissue containing bioaccumulated chemicals. The protection and maintenance of aquatic communities, fish populations, and aquatic-dependent bird and mammal populations were the ecological assessment endpoints (EAEs) evaluated. Risk questions and measurement endpoints were developed for all ROCs based on the complete and significant exposure pathways for surface water and fish tissue (for addressing risk to both fish and ROCs consuming fish) identified in the CSM.

Residual chemicals screened included pharmaceuticals; personal care products; and hormones, organobromine compounds (polybrominated diephenyl ethers [PBDEs], ethylene dibromide [EDB], and dibromochloropropane [DBCP]), and per- and polyfluoralkyl substances (PFAS). COPECs were identified by comparing the maximum concentrations of residual chemicals to conservative screening benchmarks for water. In addition, each chemical was evaluated for persistence and bioaccumulation potential based on half-lives and bioaccumulation factors, respectively. Chemicals were identified as COPECs if concentrations were greater than the screening benchmarks, or if a chemical was classified as potentially highly persistent and bioaccumulative.

ES.2 Groundwater Modeling and COPEC Refinement

A groundwater fate and transport model was developed to estimate concentrations of COPECs discharging to Woodland and McAllister Creeks over the course of 100 years of reclaimed water infiltration, beginning from present day (HDR 2021). The model output was used to refine the list of COPECs identified in the screening evaluation. For example, chemicals were removed from the list of COPECs if exposure point concentrations (EPCs) for both creeks were zero or if EPCs were less than the screening benchmark. Five COPECs were ultimately identified for quantitative risk evaluation: the surfactant 4-nonylphenol and four perfluoro surfactants (perfluoro-1-butanesulfonic acid [PFBS], perfluoro-n-hexanoic acid [PFHxA], perfluoro octanoic acid [PFOA], and perfluoropentanoic acid [PFPeA]). 4-nonylphenol was considered a surface water COPEC because the screening benchmark for water was exceeded, while the four PFAS were classified as fish tissue and wildlife COPECs due to high persistence and bioaccumulation potential.

ES.3 Exposure Analysis

The exposure analysis presents EPCs, which are estimates of the concentrations of COPECs in the creeks to which ROCs are exposed. For each COPEC, a creek-wide surface water EPC was calculated for each creek based on the maximum mass discharge (based on the 100-year groundwater fate and transport model projections) and a dilution factor (to account for the dilution of groundwater with surface water). Additionally, for the fish tissue and wildlife COPECs, fish tissue EPCs and wildlife dietary doses were calculated. Fish tissue EPCs were derived from the surface water EPCs and surface water-to-biota bioaccumulation factors (BAFs), which estimate chemical uptake into tissue from direct contact with water and dietary intake. Wildlife dietary doses were calculated for belted kingfisher and river otter using the surface water and fish tissue EPCs and species-specific food and water ingestion rates and body weights.

ES.4 Effects Characterization

The effects characterization establishes toxicity reference values (TRVs), which are toxicity thresholds below which adverse effects are not expected to occur. TRVs were derived, when possible, for surface water (for 4-nonylphenol) and fish tissue and wildlife dietary doses (for the four PFAS COPECs) using data from the scientific literature. A freshwater TRV for 4-nonylphenol was derived based on EPA guidelines for developing chronic ambient water quality criteria (AWQC). The AWQC approach uses a species sensitivity distribution (SSD) that targets a 5th percentile level of sensitivity intended to protect 95% of species in the aquatic community. A saltwater TRV for 4-nonylphenol could not be derived due to data limitations.

Fish tissue and wildlife TRVs were derived from toxicity data found in the scientific literature. Fish tissue TRVs for PFHxA and PFOA are based on no-observed-effect concentrations (NOECs) for zebrafish embryo survival and development. No data were available for PFBS or PFPeA. Bird and mammal dietary dose TRVs for PFBS (birds and mammals), PFHxA (mammals only), and PFOA (birds and mammals) are based on lowest-observed-adverse-effect levels (LOAELs) for survival, growth, and/or reproduction. No data were available for PFPeA.

ES.5 Risk Characterization

In the risk characterization, the EPCs from the exposure analysis and the TRVs from the effects characterization were used to calculate hazard quotients (HQs). HQs are used to assess potential for adverse effects. HQs greater than or equal to one indicate that there is potential for adverse effects on EAEs, and HQs less than one indicate that the potential for adverse effects causing risk to EAEs is negligible.

For 4-nonylphenol, HQs were calculated by dividing the surface water EPCs for Woodland and McAllister Creeks by the surface water TRV. For the four PFAS, HQs were based on fish tissue EPCs and wildlife dietary doses divided by their respective TRVs. All HQs are less than one (Table ES-1).

Table ES-1. Risk characterization summary

ROC	COPEC	Maximum HQ[a]	Risk Conclusion
Aquatic community	4-nonylphenol	0.00036	no unacceptable risk

ROC	COPEC	Maximum HQ[a]	Risk Conclusion
Fish	PFBS	no data	no unacceptable risk
	PFHxA	0.0000068	
	PFOA	0.00000032	
	PFPeA	no data	
Aquatic-dependent birds	PFBS	0.000000034	no unacceptable risk
	PFHxA	no data	
	PFOA	0.00000017	
	PFPeA	no data	
Aquatic-dependent mammals	PFBS	0.0000000056	no unacceptable risk
	PFHxA	0.000000012	
	PFOA	0.0000031	
	PFPeA	no data	

a Maximum for Woodland and McAllister Creeks.

COPEC – chemical of potential ecological concern
HQ – hazard quotient
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid
ROC – receptor of concern

ES.6 Summary and Conclusions

Based on their very low HQs, the potential for residual chemicals currently present in reclaimed water infiltrated into groundwater to cause risk to EAEs is negligible. Uncertainties associated with each component of the risk assessment—including COPEC selection and quantification, exposure estimation, effects estimation, and risk characterization—were evaluated and did not change the risk conclusion.

Because the ERA is based on information available at the time of writing and knowledge related to the fate, transport, and toxicity of residual chemicals is expected to increase in coming years, the ERA should be periodically updated using the most current screening and toxicity data. Updates will also be important to account for future advances in wastewater treatment technology.

1 Introduction

The LOTT Clean Water Alliance (LOTT) is a public, non-profit entity responsible for providing wastewater treatment and management for the Cities of Lacey, Olympia, and Tumwater in northern Thurston County, Washington. To meet the urban area's growing demand for wastewater management, LOTT's long-range plan relies on the production and beneficial use of reclaimed water, including the infiltration of unused reclaimed water into groundwater.

LOTT is undertaking a multi-year reclaimed water infiltration study (RWIS) to address community questions about residual chemicals[1] that might remain in reclaimed water. The RWIS is intended to evaluate whether using reclaimed water that contains traces of residual chemicals for groundwater replenishment poses risk to people and the environment. One of the RWIS tasks (Task 3) includes conducting an ecological risk assessment (ERA), which is presented in this report.

The first phase of the ERA was conducted in 2019–2020 and its results are presented in the problem formulation document (PFD) (Windward 2020). As part of the problem formulation, a screening-level risk evaluation was conducted using conservative[2] risk assumptions to identify residual chemicals of potential ecological concern (COPECs) that require further risk evaluation. This document presents results of the second phase of the ERA, in which COPECs were assessed through an exposure analysis and effects and risk characterizations.

The ERA was conducted using a standard approach in accordance with both national and regional US Environmental Protection Agency (EPA) guidance (EPA 1997a, b, 1998).

This document is organized into the following sections:

- Section 1 – Introduction
- Section 2 – Problem Formulation
- Section 3 – Groundwater Modeling Results and COPEC Refinement
- Section 4 – Exposure Analysis
- Section 5 – Effects Characterization
- Section 6 – Risk Characterization
- Section 7 – Risk Conclusions and Recommendations
- Section 8 – References

[1] Residual chemicals include pharmaceutical chemicals and chemicals found in household and personal care products.

[2] The term "conservative" is used throughout the ERA to describe assumptions related to toxicity or exposure that could contribute to an overestimation of risk expected to occur, with the purpose of being environmentally protective.

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2 Problem Formulation

This section summarizes the results of the problem formulation phase of the ERA (Windward 2020). The problem formulation included descriptions of the site, ecological setting, and conceptual site model (CSM) and identified receptors of concern (ROCs), assessment endpoints, and COPECs.

2.1 Site Description and Ecological Setting

Class A reclaimed water produced by LOTT at the Martin Way Reclaimed Water Plant (MWRWP) is infiltrated into groundwater at the Hawks Prairie Ponds and Recharge Basins site (hereinafter referred to as the Hawks prairie site); this site is located north of Lacey, Washington, between the Woodland Creek and McAllister Creek drainages, and it is the primary study area for the RWIS. At the Hawks Prairie site, reclaimed water from MWRWP is conveyed through a series of constructed wetland ponds into groundwater recharge basins, where it infiltrates through the soil into groundwater (HDR 2017c).

Groundwater in the shallow aquifer at the Hawks Prairie site flows predominantly to the southwest, with Woodland Creek being a primary point of discharge (HDR 2017a). A portion of groundwater migrates from the shallow aquifer to the sea-level aquifer, which, from the Hawks Prairie site, primarily flows toward McAllister Creek. Because the Woodland Creek and McAllister Creek watersheds are downgradient of the Hawks Prairie site, they are the focus of this ERA.

Woodland Creek flows south-to-north for approximately 11 miles through Thurston County, Washington (Figure 1). The headwaters are composed of a series of water bodies (Hicks Lake, Pattison Lake, Long Lake, Goose Pond, and Lake Lois) and form an intermittent stream until the Beatty Springs and College Creek convergence; there, the waters become a substantial perennial channel flowing northward into Henderson Inlet. Tributaries that contribute to the streamflow include College (at river mile 2.6), Eagle, Palm, Fox, Jorgensen, and Quail Creeks. The last mile of Woodland Creek is tidally influenced by Henderson Inlet.

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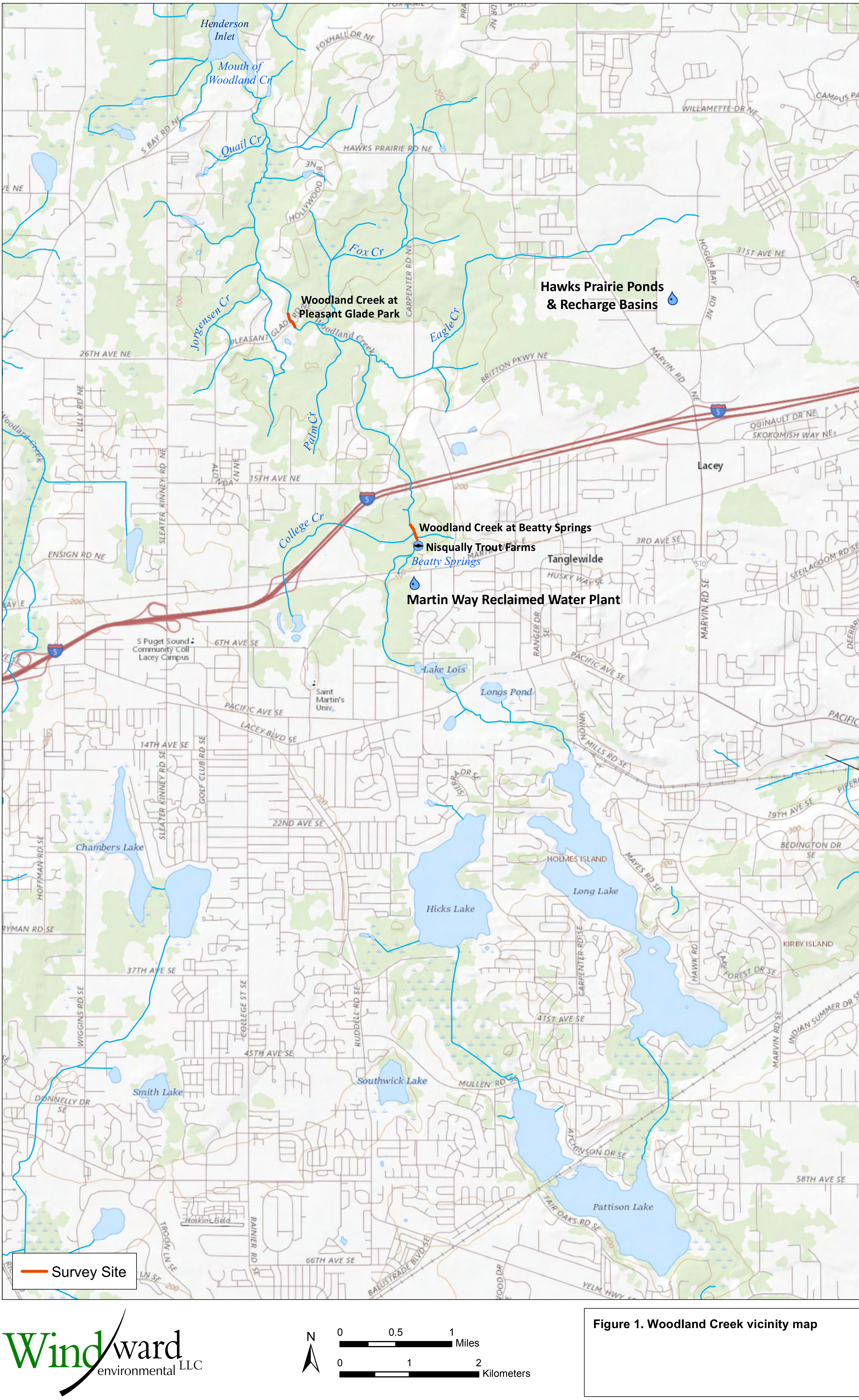

Figure 1. Woodland Creek vicinity map

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Prepared by craigh, 7/18/2019; W:\Projects\LOTT\GIS\Maps and Analyses\7076 Woodland Cr Basin.mxd

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McAllister Creek flows south-to-north for approximately 6 miles through northeast Thurston County, Washington (Figure 2). The creek is fed by a series of springs, including McAllister, Abbott, and Lodge Springs; numerous small seeps and springs along its left (west) bank; and drainage from adjacent agricultural fields and residential areas (Thurston County 1994). The entirety of McAllister Creek flows through very low-elevation areas, and the creek is tidally influenced all the way to its source. McAllister Creek discharges to the Puget Sound via a broad estuarine lagoon located within the Nisqually National Wildlife Refuge.

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N

0 0.65 1.3 Miles

0 0.65 1.3 Kilometers

Figure 2. McAllister Creek vicinity map

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Additional information about Woodland and McAllister Creeks, including habitat and associated ecological receptors, is presented in the PFD (Windward 2020). Sections of both creeks were surveyed in 2019 to confirm and supplement available information on the types of habitats, plants, benthic invertebrates, herptiles, fish, birds, mammals, and sensitive species potentially present in the two watersheds.

2.2 RECEPTORS OF CONCERN

Because of the great number and variety of species potentially residing in or utilizing the study areas, not all species were evaluated individually in the risk assessment. Instead, for aquatic species such as aquatic plants, invertebrates, fish, and herptiles that are exposed to COPECs via direct contact with water, the aquatic community was evaluated using aquatic toxicity data available for a variety of species. For birds and mammals, one receptor from each group was evaluated in the risk assessment: belted kingfisher and northern river otter were selected as ROCs to represent piscivorous species of birds and mammals, respectively.

2.3 CONCEPTUAL SITE MODEL

The ecological CSM describes the pathways by which chemicals move from surface water, tissue, sediment, and groundwater to ecological receptors in Woodland and McAllister Creeks (Figure 3). The most important exposure pathways for aquatic organisms are ingestion and direct contact. The CSM is described in more detail in Section 5 of the PFD (Windward 2020).

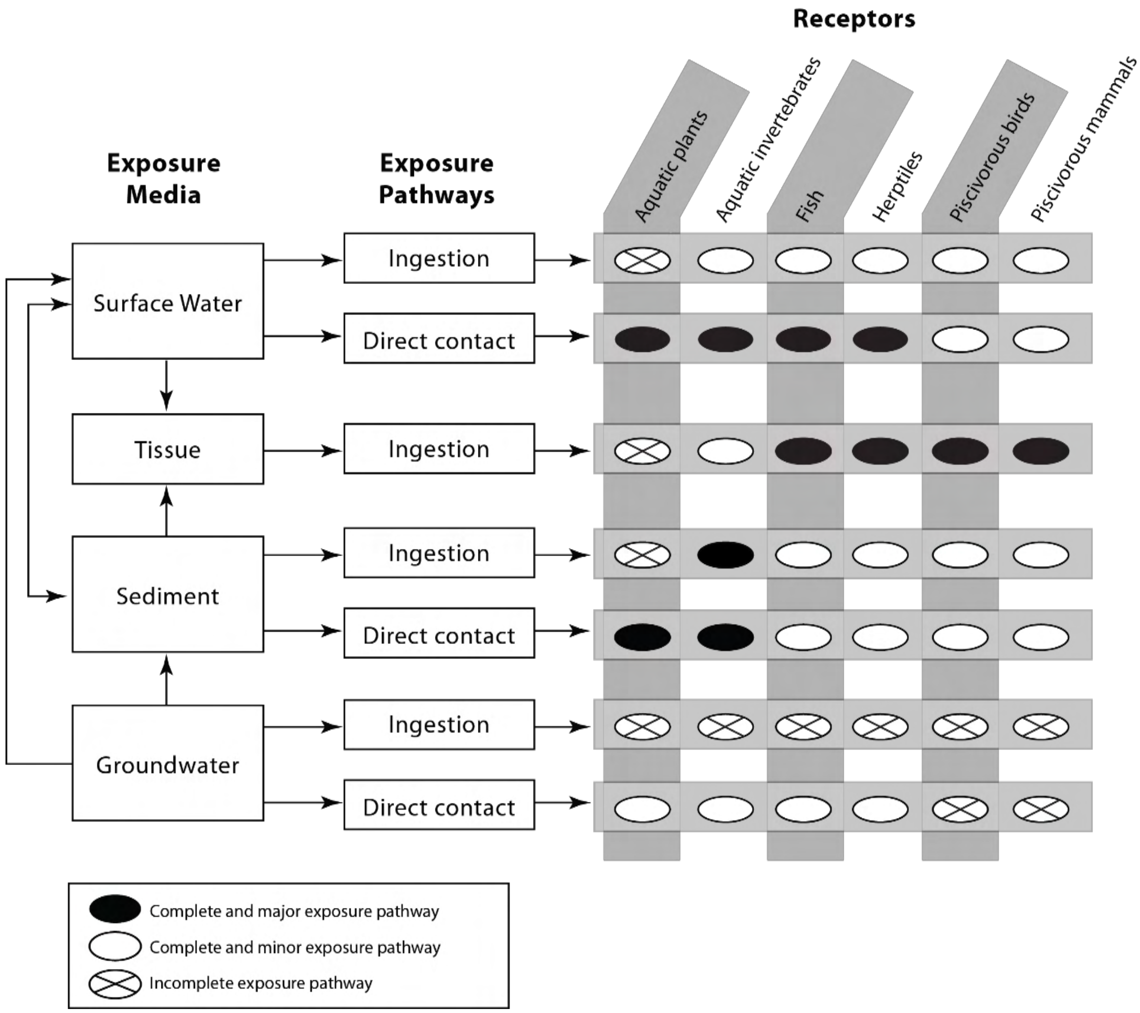

Figure 3. Exposure pathway model for Woodland and McAllister Creeks

2.4 Assessment Endpoints

The PFD outlined an approach to evaluate risks associated with major exposure pathways based on the questions and measurement endpoints presented in Table 2-1 (Windward 2020); these questions and endpoints serve as the basis for the evaluations presented in this document.

Table 2-1. Risk questions for Woodland and McAllister Creeks

Assessment Endpoint	Risk Question	Measurement Endpoint
Protection and maintenance of aquatic community populations in Woodland and McAllister Creeks	Are modeled concentrations of chemicals in surface water in Woodland and McAllister Creeks at levels that might adversely affect the aquatic community?	comparison of modeled concentrations in surface water to water TRVs[a] for aquatic species in direct contact with surface water
Protection and maintenance of fish populations in Woodland and McAllister Creeks	For chemicals that bioaccumulate, are modeled concentrations of chemicals in the tissues of fish in Woodland and McAllister Creeks at levels that might adversely affect fish populations?	comparison of modeled concentrations in fish tissue to fish tissue TRVs[b]

Assessment Endpoint	Risk Question	Measurement Endpoint
Protection and maintenance of aquatic-dependent wildlife populations in Woodland and McAllister Creeks	For chemicals that bioaccumulate, are modeled concentrations of chemicals in the tissues of prey consumed by birds and mammals in Woodland and McAllister Creeks at levels that might adversely affect aquatic-dependent wildlife populations?	comparison of calculated dietary doses for ROCs (i.e., belted kingfisher and northern river otter) to dietary dose TRVs[c]

[a] A water TRV is a concentration of a COPEC in water representing a toxicity threshold below which adverse effects are not expected to occur.

[b] A tissue TRV is a concentration of a COPEC in tissue representing a toxicity threshold below which adverse effects are not expected to occur.

[c] A dietary TRV is a dose of a COPEC (i.e., an amount ingested daily on a body weight-normalized basis) representing a toxicity threshold below which adverse effects are not expected to occur.

COPEC – chemical of potential ecological concern

ROC – receptor of concern

TRV – toxicity reference value

2.5 IDENTIFICATION OF COPECS

Residual chemicals evaluated in the screening evaluation included pharmaceuticals, personal care products, and hormones, organobromine compounds (polybrominated diephenyl ethers [PBDEs], ethylene dibromide [EDB], and dibromochloropropane [DBCP]), and per- and polyfluoralkyl substances (PFAS) (Windward 2020).[3] To identify COPECs for further evaluation in the ERA, the maximum concentration of each detected residual chemical was compared to a conservative screening-level benchmark. Each chemical was also evaluated based on its potential to be persistent and bioaccumulative. Chemicals were considered COPECs for further evaluation if they were detected in reclaimed water or porewater at concentrations greater than screening-level benchmarks, or if they were considered to be highly persistent and bioaccumulative. The list of COPECs is presented in Table 2-2.

Table 2-2. COPECs identified in the screening evaluation

COPEC	Chemical Use Category
Identified due to benchmark exceedances	
4-nonylphenol	surfactant
17-alpha ethinyl estradiol	estrogenic hormone
17-beta estradiol	estrogenic hormone
Fipronil	insecticide
Sucralose	sugar substitute
TCPP	flame retardant
TDCPP	flame retardant
Theobromine	alkaloid in chocolate and coffee

[3] Some of the residual chemicals evaluated may be associated with degradation products that were not assessed as part of the RWIS (see Section 6.1.2.2).

COPEC	Chemical Use Category
Identified based on persistence and bioaccumulation potential	
Diclofenac	anti-inflammatory
Gemfibrozil	lipid regulator
Meclofenamic acid	anti-inflammatory
PFBA	perfluoro surfactant
PFBS	perfluoro surfactant
PFHxA	perfluoro surfactant
PFNA	perfluoro surfactant
PFOA	perfluoro surfactant
PFPeA	perfluoro surfactant
Triclosan	antibacterial

COPEC – chemical of potential ecological concern
PFBA – perfluoro butanoic acid
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFNA – perfluoro-n-nonanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid
TCPP – tris(chloropropyl)phosphate
TDCPP – tris(1,3-dichloro-2-propyl)phosphate

3 Groundwater Modeling and COPEC Refinement

This section summarizes the groundwater fate and transport modeling work and the use of modeled data to refine the list of COPECs.

3.1 Groundwater Fate and Transport Modeling

Groundwater fate and transport modeling was conducted to estimate concentrations of COPECs in groundwater at certain distances from the infiltration basins, including points of discharge to Woodland and McAllister Creeks (HDR 2021). The model simulated 100 years of reclaimed water infiltration (beginning from present day), using increasing infiltration rates over time to account for potential future increases in the rate of groundwater recharge.

Groundwater concentrations at points of discharge to Woodland and McAllister Creeks were estimated based on a starting reclaimed water concentration that was modeled to disperse as groundwater moves away from the infiltration basins. The starting reclaimed water concentration was based on one of the following values:

- 95% upper confidence limit (UCL): used for chemicals with sufficient detections in reclaimed water to determine a 95% UCL via EPA's ProUCL Software
- Maximum detected value: used for chemicals with at least one detection in reclaimed water when there were insufficient detections to determine a 95% UCL
- Maximum reporting limit: used for chemicals not detected in reclaimed water (i.e., chemicals that were determined to be COPECs based on detections in porewater[4])

In addition to dispersion, attenuation (i.e., biodegradation and sorption) was factored into the model for a subset of COPECs with sufficient field data to determine a non-dispersion decay constant. Decay constants were derived for sucralose, tris(1,3-dichloro-2-propyl)phosphate (TDCPP), and triclosan based on chemical concentrations in groundwater (HDR 2021). A literature-based decay constant was applied for fipronil, which was not analyzed in groundwater but is known to degrade in the environment.

The model was used to estimate maximum daily mass loadings of COPECs to Woodland and McAllister Creeks (i.e., the sum of the maximum mass for all discharge points for each creek, representing a creek-specific total COPEC mass per day) (HDR 2021). These data were used to calculate creek-wide concentrations of COPECs, which serve as the basis of the exposure point concentrations (EPCs). EPCs are estimated

[4] Porewater data were considered uncertain and were included in the screening-level evaluation to be conservative. Only reclaimed water data were used in the groundwater fate and transport model.

concentrations of COPECs that organisms are exposed to in the creeks and serve as the basis for the exposure analysis presented in Section 4.

3.2 COPEC Refinement

The list of 18 COPECs was refined based on the results from the groundwater fate and transport model, as follows:

- Modeled EPC equals zero: EPCs for fipronil, sucralose, TDCPP, and triclosan were zero for both creeks, indicating no groundwater discharge of these chemicals into surface water due to dispersion and degradation.
- EPC less than screening-level benchmark: EPCs for tris(chloropropyl)phosphate (TCPP) and theobromine were less than the conservative benchmarks used for the problem formulation.
- Not detected in off-site groundwater: diclofenac, 17-beta estradiol, 17-alpha ethinyl estradiol, gemfibrozil, meclofenamic acid, perfluoro butanoic acid (PFBA), and perfluoro-n-nonanoic acid (PFNA) were not detected in any of the tested groundwater wells located away from the infiltration basins and upgradient of Woodland and McAllister Creeks (HDR 2017a). COPECs detected in reclaimed water but not in off-site groundwater wells were assumed to attenuate to levels less than detection limits.

With the elimination of these 13 COPECs, 5 COPECs remained for inclusion in the risk evaluation and risk characterization (Table 3-1). Of the remaining COPECs, 4-nonylphenol (hereinafter identified as the surface water COPEC) was included due to exceedances of the screening-level water benchmark, and the four PFAS (hereinafter identified as fish tissue and wildlife COPECs) were included based on persistence and bioaccumulation potential. For 4-nonylphenol, the risk characterization approach consisted of a comparison of modeled concentrations in surface water (freshwater and saltwater) to toxicity reference values (TRVs) representing threshold concentrations below which adverse effects on the aquatic community are not expected to occur.[5] For the fish tissue and wildlife COPECs, the risk characterization approach consisted of modeling concentrations in fish tissue (for evaluation of fish) and in dietary doses (for evaluation of wildlife). The modeled concentrations and doses were compared to TRVs representing threshold concentrations below which adverse effects on fish or wildlife are not expected to occur.

[5] Only a freshwater TRV could be derived (see Section 5).

Table 3-1. Refined list of COPECs for risk evaluation

COPEC	Chemical Description	Risk Characterization Approach
Surface water COPEC:		
4-nonylphenol	surfactant used in manufacturing (e.g., antioxidants, emulsifiers) and in household products (e.g., detergents); toxic to aquatic organisms; estrogenic properties	For the aquatic community, compare modeled concentrations in surface water to TRVs.
Fish tissue and wildlife COPECs:		
PFBS	perfluoro surfactants widely used in industries (e.g., electronics, chrome plating), firefighting foam, and household products (e.g., textiles, food packaging, carpet, non-stick cookware); limited ecotoxicity data; bioaccumulative in wildlife	For fish receptors, compare modeled concentrations in fish tissue to TRVs. For belted kingfisher and river otter, compare calculated dietary doses to TRVs.
PFHxA		
PFOA		
PFPeA		

COPEC – chemical of potential ecological concern
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid
TRV – toxicity reference value

3.3 Reclaimed Water Treatment Scenarios

EPCs used in the ERA are based on LOTT's current method of reclaimed water production, which uses membrane bioreactor technology to treat wastewater. Two additional reclaimed water treatment options were evaluated for potential future use (in combination with membrane bioreactor technology): reverse osmosis-advanced oxidation processes (RO-AOP) and ozone-biologically activated carbon filtration-granular activated carbon (O3-BAC-GAC) filtration. RO-AOP was determined to remove an additional 90% of 4-nonylphenol and 99% of the four PFAS, and O3-BAC-GAC was determined to remove 99% of all COPECs (Hansen 2021). EPCs that account for the use of RO-AOP and O3-BAC-GAC were calculated by HDR (2021); these EPCs are not used in the ERA.[6] The two additional treatments are introduced in the ERA for discussion purposes only.

3.4 Groundwater Model and COPEC Refinement Uncertainty

The uncertainties associated with the groundwater model, as they relate to the ERA, and the uncertainties of the COPEC refinement procedure are presented in Sections 3.4.1 and 3.4.2, respectively.

3.4.1 Groundwater fate and transport model

The general uncertainties and assumptions of the groundwater model are discussed by HDR (2021). A model uncertainty of particular importance for the ERA is that related to the attenuation factors. Non-dispersion decay constants were not used for 4-nonylphenol or PFAS, resulting in potentially high estimates of COPECs discharging

[6] EPCs used in the ERA (those based on LOTT's current reclaimed water production process) are the most conservative (i.e., highest) EPCs.

into Woodland and McAllister Creeks. Not accounting for degradation is a conservative assumption, even for highly stable chemicals such as PFAS. Based on the fate and transport model, it is estimated that the times for reclaimed water to travel from the infiltration basins to Woodland and McAllister Creeks are 46 and 13 years, respectively.[7] While degradation rates are not currently available for 4-nonylphenol or PFAS in groundwater, it is possible that degradation could occur over a period of one or more decades.

3.4.2 COPEC refinement

Seven chemicals identified as COPECs in the screening-level evaluation were removed from further consideration because they were not detected in off-site groundwater wells (Section 3.2). Two of these chemicals (17-beta estradiol and 17-alpha ethinyl estradiol) were surface water COPECs based on exceedances of screening-level benchmarks, and five (diclofenac, gemfibrozil, meclofenamic acid, PFBA, and PFNA) were fish and wildlife COPECs based on persistence and bioaccumulative potential. It was assumed that these chemicals were not detected because they attenuated to levels less than detection limits.[8] In addition, some of these chemicals are known to have short half-lives in water and may be expected to degrade in surface water and groundwater, depending on the specific environmental conditions. For example, under aerobic conditions, 17-beta estradiol and 17-alpha ethinyl estradiol have half-lives as short as 2 and 81 days, respectively, while there may be little to no decay under anaerobic conditions (Adeel et al. 2017; Ying et al. 2003).

There is some uncertainty regarding the fipronil EPCs, because fipronil was not analyzed in groundwater. A literature-based (rather than site-specific) decay rate was used in the model and applied to the maximum reclaimed water concentration.[9] Using a decay rate for fipronil is reasonable, as this chemical is known to biodegrade in anaerobic water environments (HDR 2021). Fipronil also has a high organic carbon partitioning coefficient (Koc), indicating a strong sorption/binding affinity, which is not accounted for in the model. While there is some uncertainty associated with the decay rate because it is not site-specific, the fipronil EPCs are likely conservative overall because sorption is not accounted for.

[7] It is estimated that reclaimed water introduced at the infiltration basins will first reach Woodland Creek in 2052, with estimated chemical concentrations reaching their maximum levels in 2080. Reclaimed water is estimated to have already reached McAllister Creek (in 2019), with maximum chemical concentrations estimated to occur in 2110.

[8] Removing non-detected chemicals from further evaluation is consistent with the approach taken for the screening-level evaluation, in which only detected chemicals were assessed.

[9] Insufficient data were available to calculate a 95% UCL for fipronil, so the maximum concentration was used in the model.

4 Exposure Analysis

This section describes the derivation of 1) EPCs for surface water, 2) EPCs for fish tissue, and 3) dietary doses for wildlife. The surface water EPCs were derived from the mass loading data from the groundwater fate and transport model. The fish tissue EPCs and wildlife dietary doses were modeled from the surface water EPCs. Surface water EPCs were used in this ERA as follows:

- For the surface water COPEC (identified based on exceedance of screening-level benchmarks in water), surface water EPCs represented exposure of the aquatic community through direct contact and were compared to surface water TRVs.
- For fish tissue COPECs (identified based on persistence and bioaccumulation potential), surface water EPCs were used to estimate fish tissue EPCs using bioaccumulation factors (BAFs). The fish tissue EPCs were compared to fish tissue TRVs.
- For wildlife COPECs (identified based on persistence and bioaccumulation potential), fish tissue EPCs were used as prey EPCs and surface water EPCs were used to calculate water ingestion EPCs to estimate dietary doses for aquatic-dependent wildlife. The wildlife dietary doses were then compared to dietary dose TRVs.

Surface water EPCs are presented in Section 4.1. Sections 4.2 and 4.3 describe how the surface water EPCs were used to derive fish tissue EPCs and wildlife dietary doses, respectively.

4.1 Surface Water EPCs

Surface water EPCs account for the dilution of groundwater with surface water and represent exposure on a creek-wide basis. EPCs were calculated for each COPEC and creek based on an equation reported by Einarson and Mackay (2001). The calculation uses the estimated maximum daily mass loading from the groundwater model (described in Section 3.1) and a conservative creek-specific flowrate, as shown in Equation 1:

Surface water EPC = mass discharge / flowrate **Equation 1**

Where:

Surface water EPC = estimated creek-wide concentration (ng/L)

Mass discharge = maximum daily mass loading for each creek during the 100-year modeling period (ng/day)

Flowrate = estimated water flow for each creek (L/day)

A flowrate of 489,400 L/day (0.2 cubic feet per second [cfs]) was used for Woodland Creek. This value is based on low flow at Eagle Creek in 2015 (HDR 2017b); Eagle Creek is a tributary to Woodland Creek and contains the point of maximum groundwater discharge to the Woodland Creek system. For McAllister Creek, flow data are very limited, and most available data are for McAllister Springs, which is well upstream of the portions of McAllister Creek that receive the most groundwater discharge. A flowrate of 117,456,000 L/day (48 cfs) was used for McAllister Creek based on low flow measured by the US Geological Survey between 1941 and 1949 at the Steilacoom Road gage station (Ecology 2005). Although dated, this flowrate serves as a conservative estimate because most groundwater enters McAllister Creek downstream of Steilacoom Road, where flow is expected to be greater. Mass discharges and surface water EPCs for Woodland and McAllister Creeks are presented in Table 4-1.

Table 4-1. Mass discharge and surface water EPCs

COPEC	Maximum Mass Discharge (ng/day)	EPC (ng/L)
Woodland Creek		
Surface water COPECs		
4-nonylphenol	2,444,850	5.00
Fish tissue and wildlife COPECs		
PFBS	17,540	0.0358
PFHxA	91,707	0.187
PFOA	29,835	0.0610
PFPeA	158,705	0.324
McAllister Creek		
Surface water COPECs		
4-nonylphenol	11,769,285	0.100
Fish tissue and wildlife COPECs		
PFBS	84,438	0.000719
PFHxA	441,469	0.00376
PFOA	143,622	0.00122
PFPeA	763,991	0.00650

COPEC – chemical of potential ecological concern
EPC – exposure point concentration
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid

4.2 Fish Tissue EPCs

Fish tissue EPCs were calculated for the fish tissue and wildlife COPECs (four PFAS) using the modeled surface water EPCs and surface water-to-biota BAFs. BAFs estimate chemical uptake into biota tissue from direct contact with water, as well as through dietary intake. The BAFs used in this ERA were derived from a study by Zodrow et al. (2021) that compiled data for field-based BAFs from three recent review papers (Giesy et al. 2010; Valsecchi et al. 2017; McCarthy et al. 2017). Field data are preferred to laboratory data because the former provide a measure of uptake from all exposure routes under conditions that are expected to have reached equilibrium; laboratory studies, which are relatively short-term, typically address single exposure routes.

Of the four PFAS COPECs, perfluoro octanoic acid (PFOA) had the most fish BAF field data available: 16 BAFs from 6 studies covering at least 11 species (Table 4-2). There is greater uncertainty associated with the BAFs for the other three COPECs, which had small sample sizes ranging from one to three. The geomean BAF was selected for use in estimating the fish tissue EPC for each COPEC. Uncertainties associated with the limited amount of available data and the large range in BAFs are discussed in Section 6.2.2.1.

Table 4-2. Fish tissue BAFs

COPEC	BAF[a] (L/kg dw)				Available Data	Source
	n	Minimum	Maximum	Geomean		
PFBS	3	277	6,943	916	3 studies; at least 3 species	Zodrow et al. (2021)
PFHxA	2	252	398	317	2 studies; unknown number of species	
PFOA	16	45	15,924	894	6 studies; at least 11 species	
PFPeA	1	24,273	24,273	24,273	1 study; 3 species	

[a] BAFs are based on geomean when more than one datapoint was available

BAF – bioaccumulation factor
COPEC – chemical of potential ecological concern
dw – dry weight
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid

Fish tissue EPCs were calculated using Equation 2; results are shown in Table 4-3:

$$Fish\ tissue\ EPC\ \left({}^{mg}/_{kg}\ dw\right) = \frac{Surface\ water\ EPC\ ({}^{ng}/_{L}) \times BAF\ ({}^{L}/_{kg}dw)}{10^6} \qquad \textbf{Equation 2}$$

Table 4-3. Fish tissue EPCs

COPEC	EPC (mg/kg dw)	
	Woodland Creek	McAllister Creek
PFBS	0.0000328	0.000000659
PFHxA	0.0000594	0.00000119
PFOA	0.0000545	0.00000109

COPEC	EPC (mg/kg dw)	
	Woodland Creek	McAllister Creek
PFPeA	0.00787	0.000158

COPEC – chemical of potential ecological concern
dw – dry weight
EPC – exposure point concentration
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid

4.3 WILDLIFE DIETARY DOSES

Dietary exposures to PFAS COPECs for belted kingfisher and river otter were evaluated by calculating a body weight-normalized daily dose. The components in the calculation included COPEC concentration in food, food ingestion rate, water ingestion rate, body weight, and site use factor, as shown in Equation 3:

$$\boldsymbol{Daily\ dose} = \frac{([\boldsymbol{FIR} \times \boldsymbol{EPC_{fish}}] + [\boldsymbol{WIR} \times \boldsymbol{EPC_{water}} \div \mathbf{10^6}])}{\boldsymbol{BW}} \times \boldsymbol{SUF} \qquad \textbf{Equation 3}$$

Where:

Daily dose	=	ingested dose (mg/kg bw/day)
FIR	=	food ingestion rate (kg dry weight [dw] food/day)
WIR	=	water ingestion rate (L/day)
EPC_{fish}	=	exposure point concentration in fish (mg/kg dw)
EPC_{water}	=	exposure point concentration in water (ng/L)
BW	=	body weight (kg)
SUF	=	site use factor (unitless)

Body weights used in the daily dose calculations were average male and female values from representative studies cited in EPA's *Wildlife Exposure Factors Handbook* (EPA 1993) (Table 4-4). Food ingestion rates were derived from allometric equations developed by Nagy (2001) relating body weights to ingestion rates for various classes of birds and mammals. Equations for carnivorous birds and mammals were used for belted kingfisher and river otter, respectively (Table 4-4). Water ingestion rates were derived from allometric equations for bird and mammal water intake developed by Calder and Braun (1983), as cited by EPA (1993). Although belted kingfisher and river otter may ingest small amounts of sediment incidentally while feeding, sediment data are not available for inclusion of this ingestion pathway in the daily dose calculations. The potential effect on the risk evaluation of excluding the sediment ingestion pathway is addressed in Section 4.4. The site use factor is a unitless value that represents the fraction of the diet consumed at the site relative to the fractions consumed in other areas of the receptor's home range; because this factor is unknown, a conservative value of 1 was assumed.

Table 4-4. Exposure parameter values for belted kingfisher and river otter

Parameter	Belted Kingfisher		River Otter	
	Value	Source	Value	Source
Body weight (kg)	0.148	Dunning (1984) as cited in EPA (1993): average for adult males and females	8.55	Melquist and Hornocker (1983) as cited in EPA (1993): average for adult males and females
Food ingestion rate (kg dw/day)	0.0233	Nagy (2001): equation for carnivorous birds	0.291	Nagy (2001): equation for carnivorous mammals
Water ingestion rate (L/day)	0.0164	estimated from equation reported in Calder and Braun (1983) as reported in EPA (1993)	0.683	estimated from equation reported in Calder and Braun (1983) as reported in EPA (1993)

dw – dry weight

The belted kingfisher's diet consists primarily of fish (Prose 1985; Kelly et al. 2009; Salyer and Lagler 1949; Cornwell 1963; Davis 1982), but the species has also been known to eat crustaceans, mollusks, insects, amphibians, reptiles, young birds, small mammals, and berries (White 1953; Bent 1940; Salyer and Lagler 1949). River otters are opportunistic carnivores that take advantage of the food that is most abundant and easiest to catch. Fish are their primary prey (Wise et al. 1981; Kurta 1995; Larsen 1984; Stenson et al. 1984), although they may also consume aquatic invertebrates (including crayfish, mussels, clams, and aquatic insects), frogs, snakes, and occasionally mammals and birds (Coulter et al. 1984), depending on food availability. For the purposes of this ERA, a fully piscivorous diet was assumed for both ROCs. Uncertainties associated with this assumption are addressed in Section 7.1.

Based on Equation 3, the exposure parameter values in Table 4-4 and the surface water and fish EPCs in Tables 4-1 and 4-3, respectively, were used to calculate dietary doses for belted kingfisher and river otter, as shown in Table 4-5.

Table 4-5. Bird and mammal dietary doses

COPEC	Dietary Dose (mg/kd bw/day)			
	Woodland Creek		McAllister Creek	
	Belted Kingfisher	River Otter	Belted Kingfisher	River Otter
PFBS	0.00000517	0.00000112	0.000000104	0.0000000225
PFHxA	0.00000937	0.00000204	0.000000188	0.0000000409
PFOA	0.00000859	0.00000186	0.000000172	0.0000000373
PFPeA	0.00124	0.000268	0.0000249	0.00000537

bw – body weight
COPEC – chemical of potential ecological concern
EPC – exposure point concentration
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid

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5 Effects Characterization

This section describes the effects characterization used to establish TRVs, which are chemical concentrations that represent a toxicity threshold below which adverse effects are not expected to occur.

5.1 TRVs for the Surface Water COPEC

An attempt was made to develop freshwater and saltwater TRVs for the surface water COPEC 4-nonylphenol.[10] Both freshwater and saltwater TRVs are relevant for McAllister Creek, because it is tidally influenced throughout its length. Only the freshwater TRV is relevant for Woodland Creek, because it is freshwater at the point where the greatest amount of groundwater is modeled to discharge into the creek.[11] This section describes the methods and results of the surface water TRV derivation.

5.1.1 TRV-derivation methods

Methods for deriving surface water TRVs for 4-nonylphenol are based on a species sensitivity distribution (SSD) approach that targets a 5th percentile level of sensitivity intended to protect 95% of species in the aquatic community.[12] This method generally followed EPA (1985) guidelines for chronic ambient water quality criteria (AWQC) development. Per EPA (1985), chronic criteria can be generated either directly by using chronic toxicity data or indirectly by using acute toxicity data and an acute-to-chronic ratio (ACR). The method used depends on the size of the acute and chronic toxicity datasets and the diversity of species represented. While EPA prefers that chronic criteria are calculated directly from chronic data, those data are often few in number or unavailable; therefore, the use of acute data and an ACR is a common secondary option when sufficient acute data are available but chronic data are not.

Toxicity data were compiled from EPA's ECOTOX database (EPA 2021) for tests conducted using aquatic organisms, often species and life stages known to be sensitive to pollutants (e.g., small invertebrates or fish at embryonic or larval life stages);

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[10] As discussed in Section 5.1.2, a saltwater TRV could not be derived, so only a freshwater TRV was applied.

[11] Woodland Creek is tidally influenced for only the last mile before it discharges into Henderson Inlet. The point of groundwater discharge used as the basis for the surface water EPC (i.e., where the maximum concentration of 4-nonylphenol discharges) is into a tributary, Eagle Creek, which is well upstream of Henderson Inlet.

[12] The SSD approach is more applicable for surface water COPECs than is a 20% effect level benchmark concentration (as indicated in the PFD), which is typically derived for the most sensitive species because of limited datasets. TRVs derived from 20% effect levels are more applicable for fish tissue and wildlife, which generally have data for fewer studies and species. Surface water data usually cover a relatively large number of studies and species, allowing the SSD approach to account for a wide range and distribution of species sensitivities within a natural community. The SSD approach is generally considered a higher-tier approach than the use of single benchmarks (Posthuma et al. 2019).

references for the compiled ECOTOX data are provided in Appendix A. As possible, results from each study were compared, and the toxicity test results from the most sensitive endpoint or life stage or most representative condition (e.g., longer tests or flow-through conditions) were selected (for each species). For acute tests, 48- to 96-hour EC50 (concentration that causes a nonlethal effect in 50% of an exposed population) and LC50 (concentration that is lethal to 50% of an exposed population) values were compiled for aquatic organisms; these values were augmented with additional data when data were limited.

Chronic toxicity data were compiled from tests that encompassed the life cycle of the test organism, although partial life cycle or early life stage tests for long-lived animals (e.g., fish) were also used. Acceptable chronic endpoints included survival, growth, and reproduction. Preference when selecting chronic data was given to EC20s, EC10s, and maximum acceptable toxicant concentrations (MATCs) over no-observed-effect concentrations (NOECs) and lowest-observed-effect concentrations (LOECs) when multiple endpoints were available from the same exposure.[13] Effect concentrations less than the EC10 or greater than the EC20 were not considered.[14] Toxicity data for amphibians were included as available, but aquatic plant data were not included per the EPA (1985) method.

Several other screening steps were undertaken. First, potential duplicate values were identified and removed. Duplicates were identified as toxicity endpoints that were the same and reported in different articles by the same author(s) or multiple times in the same article. The chemical purity of the active ingredient was limited to 95% or greater as a way to limit the potential effect of confounding toxicants in an exposure. Unbounded concentrations were considered only if a "greater than" operator was assigned to the toxicity endpoint; any result with a "less than" operator was excluded because such values do not lead to a conservative estimate of species sensitivity. If a lower bounded toxicity value and a higher unbounded value were available for the same species, the unbounded value was screened out.

To calculate chronic TRVs, species mean acute values (SMAVs) were first calculated as the geometric mean of the endpoint values for each species; then, for each genus, genus mean acute values (GMAVs) were calculated as the geometric mean of the SMAVs. The final acute value (FAV) was calculated as the 5th percentile GMAV. Division of the FAV by the ACR resulted in a chronic TRV.[15] This approach is consistent with EPA (1985) methods.

[13] MATCs were calculated when NOECs and LOECs were reported; an MATC is the geometric mean of the NOEC and LOEC.

[14] This excluded some values at the EC05 level as well as EC50s/LC50s reported for chronic exposures.

[15] ACRs are calculated from studies that measure both acute and chronic toxicity endpoints with the same species. The ratio of the acute toxic value to the chronic toxic value is the ACR. When selecting an ACR for criterion development (or TRV development herein), a geometric mean value of at least three ACRs (including those for both saltwater and freshwater species) is used.

In addition to the previously described data requirements, EPA (1985) lays out diversity requirements for SSDs. Specifically, for freshwater, EPA requires at least eight families to be represented in a distribution, with those families coming from various specific groups: salmonid, fish from Osteichthyes, non-salmonid chordates (e.g., amphibians), planktonic and benthic crustaceans, insects, a non-arthropod and non-chordate family (e.g., bivalves or worms), and one other family not represented by the preceding. For saltwater, the EPA-required groups include two chordate families (e.g., fish), a mysid or panaeid shrimp, a non-arthropod and non-chordate family (e.g., bivalve), three non-chordate families, and any other family. Section 5.1.2 provides greater detail about the 4-nonylphenol dataset as well as any deviations from the EPA (1985) AWQC method.

Prior to calculating TRVs for surface water, each study associated with the four most sensitive species within each SSD was critically reviewed, because the data for these species are the most important in calculating the FAV using the EPA (1985) method. In addition, the sources for ACRs were reviewed, paying special attention to the exposure conditions, species information, notes about test controls, and any other information related to data quality. Data were removed if their quality was questionable, and if a species was eliminated, studies for the next most sensitive species were reviewed until data for the four most sensitive species were deemed to be of sufficient quality. For example, any unbounded “greater than” toxicity values among the four most sensitive species in the SSD were removed as too uncertain (although such values were retained for less sensitive species).

5.1.2 TRVs for 4-nonylphenol

The following sections describe the process of deriving freshwater and saltwater TRVs for 4-nonylphenol. These TRVs are specific to 4-*para*-nonylphenol (CAS 104-40-5), rather than other commonly used formulations that include impurities (i.e., “branched” nonylphenols, CAS 84852-15-3). This approach was intended to reduce the uncertainty associated with toxicity results from tests using unclear amounts of impurities (e.g., 4-*ortho*-nonylphenol, 4-*meta*-nonylphenol, isononylphenols, octylphenols, decaphenols). As noted in Section 5.1.1, data were also limited to studies wherein the chemical purity was reported as 95% or greater.

5.1.2.1 Freshwater TRV

The freshwater 4-nonylphenol dataset from ECOTOX (EPA 2021) initially included 899 toxicity results that were filtered per the method described in Section 5.1.1; the final acute dataset included 80 unique results for 35 species and 27 genera (Appendix A, Table A1). After reviewing the literature associated with the four most sensitive species, the *Alosa sapidissima* result from Dwyer et al. (2000) was excluded because of a high level of control mortality. After reducing the SSD to 26 genera, the dataset still met the minimum acceptable diversity criteria (described in Section 5.1.1).

Chronic freshwater data were available for seven species and six genera (Appendix A, Table A2) and thus did not meet EPA's diversity requirement (i.e., eight families). Therefore, the chronic freshwater TRV for 4-nonylphenol was calculated using acute toxicity data and an ACR.

The final ACR for 4-nonylphenol is based on data compiled by EPA (2005a), the Minnesota Pollution Control Agency (MPCA 2010), and Wilkinson et al. (1997). ACRs were available for two daphnids (*Daphnia magna* [0.6586 and 6.211] and *Ceriodaphnia dubia* [9.083]), a salmonid (*Oncorhynchus mykiss* [28.11]), and a mysid (*Americamysis bahia* [8.412]); the geometric mean among these species was 8.12.[16] This mean was used to develop the chronic freshwater TRV. These data are also included in Appendix A, Table A3. ACRs used herein are based on the general group of chemicals called "nonylphenol" evaluated by EPA (2005a), rather than 4-*para*-nonylphenol specifically;[17] ACRs specific to 4-*para*-nonylphenol are not currently available.

The GMAVs, FAV, and ACR values used to derive the freshwater chronic TRV of 1.4 µg/L are shown in Table 5-1 (and Appendix A, Table A4a). The final TRV is calculated in Appendix A, Table A4b.

Table 5-1. Acute freshwater 4-nonylphenol SSD and calculated chronic TRV

Genus	Rank	GMAV (µg/L)
Dugesia	26	623
Lasmigona	25	482
Cyprinodon	24	470
Utterbackia	23	383
Daphnia	22	351
Lumbriculus	21	342
Physella	20	305
Gila	19	290
Ptychocheilus	18	256
Ceriodaphnia	17	234
Poeciliopsis	16	230
Neocaridina	15	220
Pimephales	14	219
Lepomis	13	209
Oncorhynchus	12	197
Caridina	11	195
Xyrauchen	10	173
Etheostoma	9	145

[16] The mean was calculated within species for *D. magna* before averaging among species.
[17] The SSD was focused as much as possible on the 4-*para*-nonylphenol isomer.

Genus	Rank	GMAV (µg/L)
Notropis	8	140
Oryzias	7	130
Moina	6	104
Hydra	5	96
Acipenser	4	80
Erimonax	3	80
Hyalella	2	21
Dreissena	1	7.5
FAV		**12**
Freshwater ACR		**8.12**
Chronic TRV		**1.4**

FAV – final acute value
ACR – acute-to-chronic ratio
GMAV – genus mean acute value
SSD – species sensitivity distribution
TRV – toxicity reference value

5.1.2.2 Saltwater TRV

Saltwater 4-nonylphenol data are extremely limited. The initial dataset obtained through ECOTOX (EPA 2021) contained nine toxicity values. These were filtered, according to the approach described in Section 5.1.1, to one acute value and two chronic values. The acute 96-hour LC50 of 500 µg/L was available for the copepod *Tigriopus japonicus* (Appendix A, Table A5a). Chronic values were available for the barnacle *Elminius modestus* (8-day NOEC = 10 µg/L) and the copepod *Eurytemora affinis* (10-day MATC = 10.25 µg/L) (Appendix A, Table A5b). Because of these data limitations, a saltwater TRV for 4-nonylphenol was not derived. Instead, the final freshwater TRV of 1.4 µg/L will be applied to saltwater conditions. Uncertainties associated with the use of a freshwater 4-nonylphenol TRV are discussed in Section 6.1.2.4.

5.1.3 Summary of surface water TRV development

The final chronic freshwater TRV for 4-nonylphenol was calculated as 1,400 ng/L and was based on the ECOTOX database (EPA 2021). The final dataset (after filtering the data as described in Section 5.1.1) included 26 genera. An ACR of 8.12 was used to convert the 5th percentile of SMAVs to the chronic TRV. Because a comparable value could not reasonably be derived for saltwater data, this TRV was applied to both Woodland and McAllister Creeks, regardless of water salinity. Uncertainties associated with this TRV are discussed in Section 6.1.2.4.

5.2 TRVs FOR FISH TISSUE COPECs

Fish tissue TRVs are concentrations of COPECs in tissue that represent a toxicity threshold below which adverse effects are not expected to occur in fish. A literature search was conducted in May 2021 for scientific publications that could be used for the development of TRVs for the four fish tissue PFAS COPECs. The search used EPA's ECOTOX database (EPA 2021), the US National Library of Medicine's TOXLINE database (NCBI 2021), and Google Scholar to identify publications presenting fish tissue concentrations associated with adverse effects on fish growth, reproduction, or survival.

No data were found for perfluoro-1-butanesulfonic acid (PFBS) or perfluoropentanoic acid (PFPeA). Only one study was found that related fish tissue concentrations to effects on fish (Gaballah et al. 2020). This study exposed zebrafish embryos to a range of PFOA and perfluoro-n-hexanoic acid (PFHxA) concentrations, in separate exposures, immediately after fertilization. After six days, no effects on survival or development were observed. Tissue concentrations associated with the highest exposure concentrations are presented in Table 5-2. These NOECs for tissue are used as TRVs, because data are not available to derive effect levels associated with a 20% reduction in growth, reproduction, or survival. Because these TRVs are levels at which no effects were observed, they are conservative values. Uncertainties associated with the lack of fish tissue data are discussed in Section 6.2.2.2.

Table 5-2. Fish tissue TRVs

COPEC	TRV (mg/kg dw)	Effect Description	Source
PFBS	no data	na	na
PFHxA	8.77	NOEC for survival and development of zebrafish embryos	Gaballah et al. (2020)
PFPeA	no data	na	na
PFOA	171	NOEC for survival and development of zebrafish embryos	Gaballah et al. (2020)

COPEC – chemical of potential ecological concern
dw – dry weight
na – not applicable
NOEC – no observed effect concentration
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid
TRV – toxicity reference value

5.3 TRVS FOR WILDLIFE COPECS

Wildlife TRVs are dietary doses of COPECs that represent a toxicity threshold below which adverse effects are not expected to occur. The wildlife TRVs used in this ERA were derived primarily from data compiled by Zodrow et al. (2021). The authors conducted a thorough review of the scientific literature for wildlife toxicity data for a comprehensive list of PFAS compounds, including the four PFAS wildlife COPECs evaluated in this ERA. The literature search was conducted using 1) multiple search engines, 2) 11 key recent US guidance documents, and 3) EPA's ECOTOX database. Because Zodrow et al. (2021) submitted for publication in 2020 and conducted the associated ECOTOX search in February 2019, Windward Environmental LLC (Windward) conducted a search in May 2021 for literature published in 2019 or later to identify papers that may not have been included by Zodrow et al. (2021). The additional search was conducted using ECOTOX (EPA 2021) and the US National Library of Medicine's TOXLINE database (NCBI 2021). The only additional paper identified in the Windward search was a bird study evaluating the effects of PFOA (Bursian et al. 2021).

Both the Zodrow et al. (2021) and Windward literature searches were limited to studies that were three days or longer and that reported on endpoints related to growth, reproduction, or survival. Only studies with dosing based on the ingestion of food or water or oral gavage were included; other forms of exposure, such as subcutaneous implant or injection, were not considered environmentally relevant pathways.

Zodrow et al. (2021) derived no-observed-adverse-effect levels (NOAELs) and lowest-observed-adverse-effect levels (LOAELs) using EPA's step-wise methodology for deriving NOAEL- and LOAEL-based TRVs for ecological soil screening levels (Eco-SSLs) (EPA 2005b). The LOAEL-based TRVs identified by Zodrow et al. (2021) and derived from Bursian et al. (2021) (Table 5-3) are used in this ERA, rather than effect doses associated with a 20% reduction of growth, reproduction, or survival (ED20s). The derivation of ED20s would involve modeling the dose-response data from each study. The LOAEL-based TRVs are several orders of magnitude less than the estimated dietary doses in Table 4-5; therefore, the additional modeling for ED20s was not conducted because their use would not result in different risk conclusions (i.e., hazard quotients [HQs] < 1).

Table 5-3. Wildlife dietary TRVs

COPEC	No. of Studies	Species	Endpoints Evaluated	LOAEL-based TRV (mg/kg bw/day)	Source
Birds					
PFBS	3	bobwhite quail, mallard	survival, growth, reproduction	153	Zodrow et al. (2021)
PFHxA	no data	na	na	na	na
PFOA	1	Japanese quail	growth	52	Bursian et al. (2021)
PFPeA	no data	na	na	na	na
Mammals					
PFBS	6	rat, mouse	survival, growth, reproduction	200	Zodrow et al. (2021)
PFHxA	3	rat, mouse	survival, growth, reproduction	175	Zodrow et al. (2021)
PFOA	> 25	rat, mouse	survival, growth, reproduction	0.6	Zodrow et al. (2021)
PFPeA	no data	na	na	na	na

bw – body weight
COPEC – chemical of potential ecological concern
LOAEL – lowest observed adverse effect level
na – not applicable
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid
TRV – toxicity reference value

Sufficient data were available to allow for the derivation of LOAEL-based TRVs for PFBS and PFOA for both birds and mammals, and for PFHxA for mammals (Table 5-3). A limited number of studies were available for the derivation of most TRVs, with the exception of PFOA for mammals, for which there were more than 25 studies. Uncertainties associated with the lack of data for some compounds and the limited amount of data for others are discussed in Section 6.3.2.2.

6 Risk Characterization

The risk characterization addresses the assessment endpoints, risk questions, and measurement endpoints for aquatic organisms and aquatic-dependent wildlife identified in Table 2-1. An HQ approach is used to characterize the potential for adverse effects by comparing EPCs or daily doses to TRVs according to Equation 4:

$$\boldsymbol{HQ} = \frac{(\boldsymbol{EPC\ or\ Dose})}{\boldsymbol{TRV}} \quad \textbf{Equation 4}$$

The HQ indicates the factor by which the EPC or dose is above or below the toxicity threshold. HQs less than one indicate low potential for adverse effects. HQs greater than one indicate the potential for unacceptable effects. If any HQs are greater than one, further evaluation is necessary to understand the ecological significance of the potential effects, such as the spatial extent of unacceptable effects and whether the estimated effects are limited to individual organisms or extend to the entire population. The interpretation of HQs must also consider the uncertainties associated with exposure and effects data.

The following sections present the risk characterization results for the three assessment endpoints addressed in this ERA.

6.1 Aquatic Community Endpoint

This section presents the surface water HQs, a consideration of HQ uncertainties, and a summary of risk characterization results for the assessment endpoint addressing the protection and maintenance of aquatic community populations in Woodland and McAllister Creeks.

6.1.1 Surface water HQs

The HQs for 4-nonylphenol in Woodland and McAllister Creeks are all well below one (Table 6-1). Data were not available to calculate an HQ for 4-nonylphenol in saltwater.

Table 6-1. Surface water HQs

	Surface Water HQ		
	Woodland Creek	McAllister Creek[a]	
COPEC	Freshwater	Freshwater	Saltwater
4-nonylphenol	0.00036	0.000072	no data

[a] Freshwater and saltwater HQs are relevant for McAllister Creek because it is tidally influenced throughout its length. Woodland Creek is freshwater at the point where maximum concentrations of COPECs in groundwater discharge to the creek, based on the fate and transport model.

COPEC – chemical of potential ecological concern

HQ – hazard quotient

6.1.2 Uncertainty evaluation

This section discusses uncertainty associated with additive effects, degradation products, exposure point concentrations, and TRVs used in the aquatic community risk characterization.

6.1.2.1 Additive effects uncertainty

Effects caused by additive interactions of residual chemicals were not factored into the effects assessment. Generally, it can be assumed that toxicity is additive if the individual chemicals induce the same health effects by similar modes of action (MOAs) (EPA 2000a). There is, however, substantial uncertainty when assessing risk from mixtures, due primarily to the lack of toxicological data, particularly with three or more chemicals (EPA 2000b). Additionally, the diversity of the chemicals in the mixture, their reactions in the ambient environment, and their interactions with one another increase the uncertainty associated with predicting mixtures toxicity.

For example, PFAS include thousands of heterogenous chemicals, but there is substantial toxicological understanding for only a few (Ankley et al. 2021). Furthermore, these chemicals "often enter the environment as poorly characterized complex mixtures of parent molecules and their precursors, degradation products, and metabolites." However, while agents such as dioxins have a known molecular initiating event (MIE) (Van den Berg et al. 1998), the MIE and even the more general MOA are not understood for PFAS; it is unknown if these effects are shared for subsets of PFAS (Ankley et al. 2021).

Recent efforts to develop MOA classification in chemical risk assessment through existing MOA frameworks have yielded contradictory results when applied across a variety of chemicals, indicating that more work is needed to harmonize and update these systems before they can be applied to improve risk assessments (Kienzler et al. 2017). Thus, the ability to account for interactive effects through a MOA approach for most contaminants of emerging concern is currently very limited, and the effects of potential additive, antagonistic, or synergistic interactions of residual chemicals on risk predictions are unknown.

6.1.2.2 Degradation products uncertainty

The residual chemicals evaluated as part of the RWIS may have associated degradation products for which toxicity was not assessed as part of the ERA. While concentrations of degradation products are often less than those of the parent compound, toxicity can be less or greater than that of the chemical of origin. Thus, the effects of degradation products are unknown; however, based on the low risk associated with the chemicals evaluated, the toxicity of degradation products would need to be substantially greater than that of the parent chemical to pose unacceptable risk.

6.1.2.3 Exposure point concentration uncertainty

Several of the factors used to calculate surface water EPCs likely contributed to higher (i.e., more conservative or overestimated) EPCs for Woodland and McAllister Creeks. First, as discussed in Section 3.4.1, decay constants were not applied in the groundwater model for any of the COPECs evaluated. Second, in considering the dilution of groundwater with surface water, the maximum (instead of the average) daily mass loading for each creek was used. In addition, the flowrates for both creeks are very conservative, in that they are based on data collected during the dry season for upstream (i.e., low-flow) parts of the creek. This is true despite the additional uncertainty in the McAllister Creek flowrate, which uses dated information.

6.1.2.4 TRV uncertainty

General Uncertainty

Surface water TRVs are derived from toxicity tests conducted in the laboratory, which might not accurately reflect field conditions in the environment of interest. Some example field conditions include the potential natural adaptation of communities to toxicants and the potential for parameters such as pH or organic carbon to affect chemical fate and bioavailability. It is not known if conditions in the laboratory are more or less toxic than those in the field; therefore, it is not possible to determine whether the 4-nonylphenol TRV derived from laboratory data is likely to over- or underestimate risk.

Similarly, water toxicity studies are often conducted with model test organisms, which are intended to represent broad groupings of species. This approach results in TRVs for one species being applied to a different species, which may be more or less sensitive to the COPEC. To address uncertainties, the surface water TRV for 4-nonylphenol is based on the most sensitive species among those with available toxicity test data. While this may result in a higher estimate of ecological risk for less sensitive species, the approach is conservative and appropriate for the most sensitive species.

4-nonylphenol TRV Uncertainty

The freshwater TRV for 4-nonylphenol (Section 5.1.2) was developed using an acute SSD and an ACR. When there are relatively few chronic toxicity data, as is the case for 4-nonylphenol in freshwater, EPA (1985) allows for the ACR approach as an alternative to developing a chronic SSD if acute-to-chronic data are available for at least three species. While small SSDs have questionable representativeness, there is also uncertainty with the ACR approach when the ACR dataset is small and/or variable. The four ACRs for 4-nonylphenol range from 0.6586 to 28.11 for different species (Appendix A, Table A3), and even within species (i.e., for *D. magna*) there is an order of magnitude difference (0.6586 versus 6.211). It is not known if a larger dataset would result in a higher or lower ACR than the value of 8.12 used to derive the TRV (Table 5-1); however, the ACR would need to be at least an order of magnitude higher for HQs to exceed one in Woodland Creek and at least three orders of magnitude higher for HQs to exceed one in McAllister Creek.

A saltwater TRV could not be developed because of the lack of data for 4-nonylphenol. Instead, the freshwater TRV was applied. In order to evaluate the uncertainty associated with applying a freshwater TRV to saltwater species, an additional literature search was conducted. In a study by Lussier et al. (2000), an acute saltwater AWQC of 12.4 µg/L was derived for 4-nonylphenol (relying on the 4-nonylphenol branched formulation [Chemical Abstracts Service No. 84852-15-3], which was not considered herein).[18] Dividing 12.4 µg/L by the ACR of 8.12 would result in a chronic saltwater TRV of 1.5 µg/L, a value very similar to the freshwater chronic TRV of 1.4 µg/L (Table 5-1). Therefore, the calculated 4-nonylphenol freshwater HQs for McAllister Creek would be almost identical to the saltwater HQs calculated using the saltwater TRV based on the 4-nonylphenol, branched toxicity data.

Representativeness of TRVs for Threatened and Endangered Species

Three species of fish (all salmonids) that may be present within the aquatic communities of Woodland or McAllister Creeks are designated as threatened by the US Fish and Wildlife Service (USFWS): bull trout, Chinook salmon, and winter steelhead trout.[19] No fish species are listed as endangered.[20] The SSD for 4-nonylphenol contains the genus Oncorhynchus, which is in the salmon family (and includes both Chinook salmon and steelhead trout); therefore, the TRV based on this SSD should be protective of salmonids.

6.1.3 Aquatic community risk characterization summary

Surface water HQs for 4-nonylphenol are more than three orders of magnitude less than one, based on freshwater TRVs for both Woodland and McAllister Creeks. Although a saltwater TRV could not be developed for 4-nonylphenol because of the lack of data, the limited available data indicate that toxicity values for saltwater organisms are within the range of those for freshwater organisms (see Section 5.1.2), which would result in similar HQs for saltwater organisms in McAllister Creek. The low HQs indicate a negligible potential for adverse effects on the aquatic populations in Woodland and McAllister Creeks from exposure to 4-nonylphenol in surface water. Uncertainties associated with EPCs and TRVs are unlikely to result in HQs exceeding one.

[18] As described in Section 5.1.2, the derivation of the 4-nonylphenol TRV was based on the para isomer rather than on data for more uncertain mixtures of para, ortho, and meta isomers and deca- or octophenols and isophenols, all of which can occur as impurities in nonylphenol products.

[19] Appendix B of the PFD (Windward 2020) also lists the following fish species as threatened: 1) chum salmon, which are listed as threatened for only the Hood Canal breeding population, and 2) sockeye salmon, which are listed as threatened in only Lake Ozette (Washington) and Snake River (Idaho).

[20] White sturgeon is listed as endangered, but the listing is specific to the Kootenai subspecies in Idaho, Montana, and British Columbia.

6.2 FISH ENDPOINT

This section presents the fish tissue HQs, a consideration of HQ uncertainties, and a summary of risk characterization results for the assessment endpoint addressing the protection and maintenance of fish populations in Woodland and McAllister Creeks.

6.2.1 Fish Tissue HQs

The HQs for fish tissue are all well below one (Table 6-2). No data were available to calculate HQs for PFBS or PFPeA.

Table 6-2. Fish tissue HQs

COPEC	Fish tissue NOAEL HQ	
	Woodland Creek	McAllister Creek
PFBS	no data	no data
PFHxA	0.0000068	0.00000014
PFOA	0.00000032	0.0000000064
PFPeA	no data	no data

COPEC – chemical of potential ecological concern
HQ – hazard quotient
NOAEL – no-observed-adverse-effect level
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid

6.2.2 Uncertainty evaluation

This section discusses uncertainty in the fish risk characterization associated with EPCs and TRVs. Uncertainties associated with additive effects and degradation products, as discussed in Sections 6.1.2.1 and 6.1.2.2, also apply to the fish risk characterization.

6.2.2.1 Exposure point concentration uncertainty

EPCs for fish tissue COPECs were based on BAFs (Table 4-2), which were limited in two ways. First, three or fewer BAFs were available in the literature for PFBS, PFHxA, and PFPeA, and second, the range of BAFs was large (more than one order of magnitude) for PFBS and PFOA; the EPC for the latter was based on 16 BAFs, representing the largest range. However, use of the highest BAF for PFOA (Table 4-2), rather than the geomean, results in a tissue concentration of 976 ng/kg in Woodland Creek and an HQ of 0.0000057, indicating that the large range of BAFs does not affect the risk conclusions.

6.2.2.2 TRV uncertainty

General Uncertainty

A general uncertainty associated with the toxicity studies used to derive the fish tissue TRVs is whether the effects observed in a controlled laboratory environment would be observed to a greater, lesser, or similar extent in the natural environment. The sensitivity of fish to PFAS compounds would need to be several orders of magnitude higher in the natural environment than in the laboratory for HQs to exceed one. There is no ecotoxicological or ecological basis for concern that field sensitivity would exceed laboratory sensitivity by such a wide margin.

TRV Derivation Uncertainty

The primary uncertainty associated with the TRVs derived for fish tissue is that limited data are available. Only one study relating fish tissue concentrations of PFAS compounds to effects on fish was identified (Gaballah et al. 2020); this study did not include data on PFPeA or PFBS. In addition, this study was based on a single species (zebrafish), so there is uncertainty regarding how representative the TRVs are for other types of fish.

Other uncertainties specific to the study by Gaballah et al. (2020) are:

- No effects were observed at the highest exposure concentrations tested, so the TRVs are based on NOECs, which represent conservative values. The higher concentrations at which effects might be observed (i.e., LOECs) are not known.
- The fish tissue concentrations are based on fish embryos. Chemical concentrations in embryos and adults can vary depending on the chemical, and data could not be found regarding concentrations of PFOA and PFHxA in embryos relative to concentrations in adults. Therefore, there is uncertainty regarding how representative the TRVs are for adult fish.

Because of the limited amount of available data, the TRVs derived for fish tissue are considered highly uncertain. However, the TRVs would need to be several orders of magnitude lower than those presented in Table 5-2 for HQs to exceed one.

Representativeness of TRVs to Threatened and Endangered Species

As noted in Section 6.1.2.4, three species of fish that may be present within the aquatic communities of Woodland or McAllister Creeks are designated as threatened by the USFWS: bull trout, Chinook salmon, and winter steelhead trout, all of which are salmonids. Toxicity data for the four PFAS COPECs in fish tissue were found for only zebrafish, and data on the sensitivity of zebrafish relative to the sensitivity of salmonids were not available. However, TRVs for salmonids would need to be several orders of magnitude higher than the TRVs for zebrafish used in this evaluation in order for HQs to exceed one. To evaluate this uncertainty, a quantitative evaluation of toxicity values for many species and chemicals was conducted using the underlying data from EPA's WebICE tool (EPA 2017), which uses regression models to predict toxicity in species with little or no empirical toxicity data. A review of data in the WebICE database indicates that it is highly unlikely that toxicity values for fish would range over five orders of magnitude. Fish acute water toxicity data are available for 629 chemicals (not including the COPECs); no chemical is associated with fish toxicity data that range over 5 orders of magnitude (maximum of 3.7).

6.2.3 Fish risk characterization summary

HQs for PFOA and PFHxA in fish tissue are more than five orders of magnitude less than one. HQs could not be calculated for PFBS or PFPeA because of the lack of data to derive TRVs; however, these TRVs would need to be several orders of magnitude higher than those derived for PFOA and PFHxA for HQs to exceed one. In general, available data suggest that the toxicity of a shorter-carbon-chain-length PFAS, such as PFBS, is less than the toxicity of longer-carbon-chain compounds such as PFOA (Ankley et al. 2021), so there is reason to expect that fish tissue HQs for PFBS and PFPeA would be less than those for PFOA and PFHxA. These data indicate a negligible potential for adverse effects on the fish populations in Woodland and McAllister Creeks from the bioaccumulation of PFOA, PFHxA, PFBS, and PFPeA in fish tissue. Although there are uncertainties associated with the BAFs and TRVs, they are unlikely to affect risk conclusions.

6.3 AQUATIC-DEPENDENT WILDLIFE ENDPOINT

This section presents the bird and mammal HQs, a consideration of HQ uncertainties, and a summary of risk characterization results for the assessment endpoint addressing the protection and maintenance of aquatic-dependent wildlife populations in Woodland and McAllister Creeks.

6.3.1 Bird and mammal dietary HQs

The wildlife dietary dose HQs are all well below one (Table 6-3). No data were available to calculate HQs for PFHxA (birds) or PFPeA (birds and mammals).

Table 6-3. Bird and mammal dietary HQs

	Dietary LOAEL HQ			
	Woodland Creek		McAllister Creek	
COPEC	Bird	Mammal	Bird	Mammal
PFBS	0.000000034	0.0000000056	0.00000000068	0.00000000011
PFHxA	no data	0.000000012	no data	0.00000000023
PFOA	0.00000017	0.0000031	0.0000000033	0.000000062
PFPeA	no data	no data	no data	no data

COPEC – chemical of potential ecological concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PFBS – perfluoro-1-butanesulfonic acid
PFHxA – perfluoro-n-hexanoic acid
PFOA – perfluoro octanoic acid
PFPeA – perfluoropentanoic acid

6.3.2 Uncertainty evaluation

This section discusses uncertainty associated with dietary exposure estimation and TRVs in the aquatic-dependent wildlife risk characterization. Uncertainties associated with additive effects and degradation products, as discussed in Sections 6.1.2.1 and 6.1.2.2, also apply to the wildlife risk characterization.

6.3.2.1 Dietary exposure uncertainty

Data are not available for PFAS compounds concentrations in Woodland or McAllister Creek sediment, so it was not possible to characterize the contributions of incidental sediment ingestion to the wildlife dietary doses. Incidental sediment ingestion is estimated to be a small portion of the diet for belted kingfisher and river otter (less than 5%), based on information for similar wildlife species (Beyer et al. 1994; Beyer et al. 2008). Therefore, the additional doses of PFAS compounds from ingested sediment will have only a minimal effect on the extremely low HQs.

The fish EPCs are based on BAFs, which are limited in their availability and have a large range, as discussed in 6.2.2.1. However, given the very low wildlife HQs (Table 6-3), these uncertainties have a minimal effect on the HQs. For example, while use of the highest BAF for PFOA rather than the geomean increases the PFOA HQ by approximately two orders of magnitude, the highest PFOA HQ would still be only be 0.000055.

While it is assumed that the belted kingfisher and river otter diets consist entirely of fish, they may also consume relatively small amounts of benthic invertebrates and other organisms (see Section 4.3). Based on data presented by Zodrow et al. (2021), benthic invertebrate geomean BAFs for the PFAS COPECs are within approximately one to two orders of magnitude of the fish geomean BAFs, indicating that including modeled benthic invertebrates in the wildlife diets would not have a substantial effect on the extremely low HQs.

6.3.2.2 TRV derivation uncertainty

General Uncertainty

A general uncertainty associated with the bird and mammal toxicity studies used to derive the wildlife TRVs is whether the effects observed in a controlled laboratory environment would be observed to a greater, lesser, or similar extent in the natural environment. Wildlife dietary doses of PFAS compounds associated with adverse effects would need to be at least five orders of magnitude less in the natural environment than in the laboratory for any of the HQs to exceed one. While wildlife (i.e., bird and mammal) toxicity data are not available through EPA's WebICE database (EPA 2017), fish and invertebrate data are available for hundreds of chemicals (629 and 585, respectively); amphibian data are also available for a smaller number of chemicals (37). Among all chemicals (none of which were the COPECs) and species with available data, only a single chemical had a range of toxicity values exceeding five orders of magnitude.

TRV Derivation Uncertainty

A primary uncertainty associated with TRVs for wildlife is the limited toxicity data that are available for many chemicals. For example, no data were found for PFPeA, and no bird diet toxicity data were found for PFHxA. The only bird species used in the PFAS toxicity tests were bobwhite quail, Japanese quail, and mallard, and the only mammal species were rat and mouse. It is unknown whether the extrapolation of results from this limited variety of laboratory species to belted kingfisher and river otter would contribute to an over- or underestimation of the TRV, because data are not available to determine if the receptor species are more or less sensitive than the tested species. However, it is highly unlikely that toxicity data for other PFAS COPECs or other species would be different enough from the available data to result in HQs exceeding one.

Representativeness of TRVs to Threatened and Endangered Species

One bird species, marbled murrelet, is the only aquatic-dependent wildlife species potentially using Woodland or McAllister Creeks that is designated as threatened by the USFWS; no aquatic-dependent wildlife species that might use the site are endangered. Data are not available to indicate how representative bobwhite quail, Japanese quail, and mallard are of marbled murrelet. However, marbled murrelet would need to be at least five orders of magnitude more sensitive than the laboratory species in order for HQs to exceed one. EPA's WebICE data were evaluated to understand reasonable ranges of sensitivities among fish, invertebrate, and amphibian species (EPA 2017); no analogous wildlife data were available. Evaluation of the non-wildlife data found that ranges of toxicity values tended not to be much less than five orders of magnitude among species and across hundreds of (non-COPEC) chemicals. This result suggests that inter-specific differences in sensitivity are unlikely to be large enough for HQs to exceed one.

6.3.3 Aquatic-dependent wildlife risk characterization summary

HQs for PFBS, PFHxA, and PFOA based on estimated dietary doses are more than five orders of magnitude less than one. HQs could not be calculated for PFPeA because of the lack of data to derive a TRV; however, the TRV would need to be several orders of magnitude higher than those derived for the other PFAS COPECS in order for HQs to exceed one. The extremely low HQs indicate negligible potential for adverse effects on the wildlife populations consuming aquatic organisms from Woodland and McAllister Creeks. The uncertainties associated with the wildlife HQ calculations are unlikely to affect risk conclusions based on the very low HQs.

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7 Summary and Conclusions

The scope of the RWIS included evaluating ecological risk from chemicals of emerging concern, such as pharmaceuticals and personal care products. The ERA was conducted in accordance with EPA guidelines and based on screening data available in 2019 for the problem formulation and toxicity data available in 2021 for the effects and risk characterizations. The list of COPECs identified in the problem formulation was refined based on exposure data from the groundwater fate and transport model for Woodland and McAllister Creeks, resulting in detailed risk evaluations for five chemicals: one surface water COPEC (4-nonylphenol) and four fish tissue and wildlife COPECs (PFBS, PFHxA, PFOA, and PFPeA). The surface water COPEC was evaluated for the aquatic community based on direct exposure to surface water. Fish tissue and wildlife COPECs were evaluated, due to high persistence and bioaccumulation potential, by modeling fish tissue concentrations and aquatic-dependent wildlife dietary doses. The potential for adverse effects on aquatic community populations, fish populations, and aquatic-dependent wildlife populations was determined to be negligible based on comparisons of EPCs or dietary doses to TRVs; all HQs were orders of magnitude less than one. Uncertainties associated with each component of the risk assessment—including COPEC selection and quantification, exposure estimation, effects estimation, and risk characterization—were evaluated and determined to have a low impact on the risk conclusions. It is important to note that the exposure scenarios evaluated in the ERA represent potential future conditions that do not currently occur and may not occur for several decades. Chemical concentrations were assumed to stay consistent for the duration of the 100-year groundwater fate and transport model run; in reality, the types and concentrations of chemicals in reclaimed water will change over time, as chemicals are phased in and out and wastewater treatment technologies advance.

Applying either of the other reclaimed water treatment options discussed in Section 3.3 (RO-AOP or O3-BAC-GAC) would result in 90 to 99% lower surface water EPCs and thus even lower HQs than those based on current reclaimed water production methods. If COPEC concentrations were to increase in the future to the extent that HQs approached one, either of the additional treatments would provide sufficient removal to be ecologically protective.

There are data gaps and uncertainties in the ERA, many of which are due to the emergent status of the chemicals evaluated. Because the ERA is based on information available at the time of writing and knowledge related to the fate, transport, and toxicity of chemicals of emerging concern is expected to increase in coming years, the ERA should be periodically updated using the most current screening and toxicity data. The limitations and uncertainties of the ERA should be considered when applying the results to wastewater management decisions.

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