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Carli Creek Regional Water Quality Project: Assessing Water Quality Improvement at an Urban Stormwater Constructed Wetland

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Executive Summary

Stormwater management is an ongoing challenge in the United States and the world at-large. As state and municipal agencies grapple with conflicting interests like encouraging land development, complying with permits to control stormwater discharges, "urban stream syndrome" effects, and charges to steward natural resources for the long-term, some agencies may turn to constructed wetlands (CWs) as aesthetically pleasing and functional natural analogs for attenuating pollution delivered by stormwater runoff to rivers and streams. Constructed wetlands retain pollutants via common physical, physicochemical, and biological principles such as settling, adsorption, or plant and algae uptake. The efficacy of constructed wetlands for pollutant attenuation varies depending on many factors such as flow rate, pollutant loading, maintenance practices, and design features. In 2018, the culmination of efforts by Clackamas Water Environment Services and others led to the opening of the Carli Creek Water Quality Project, a 15-acre constructed wetland adjacent to Carli Creek, a small, 3500-ft tributary of the Clackamas River in Clackamas County, OR. The combined creek and constructed wetland drain an industrialized, 438-acre, impervious catchment. The wetland consists of a linear series of a detention pond and three bioretention treatment cells, contributing a combined 1.8 acres of treatment area (a 1:243 ratio with the catchment) and 3.3 acre-feet of total runoff storage. In this study, raw pollutant concentrations in runoff were evaluated against International Stormwater BMP database benchmarks and Oregon Water Quality Criteria. Concentration and mass-based reductions were calculated for 10 specific pollutants and compared to daily precipitation totals from a nearby precipitation station.

Mass-based reductions were generally higher for all pollutants, largely due to runoff volume reduction on the treatment terrace. Concentration-based reductions were highly variable,

and suggested export of certain pollutants (e.g., ammonia), even when reporting on a mass-basis. Mass load reductions on the terrace for total dissolved solids, nitrate+nitrite, dissolved lead, and dissolved copper were 43.3 ± 10%, 41.9 ± 10%, 36.6 ± 13%, and 43.2 ± 16%, respectively. *E. coli* saw log-reductions ranging from -1.3 —3.0 on the terrace, and -1.0 1.8 in the creek. Oregon Water Quality Criteria were consistently met at the two in-stream sites on Carli Creek for *E. coli* with one exception, and for dissolved cadmium, lead, zinc, and copper (with one exception for copper). However, dissolved total solids at the downstream Carli Creek site was above the Willamette River guidance value 100 mg/L roughly 71% of the time.

The precipitation record during the study was useful for explaining certain pollutant reductions, as several mechanisms are driven by physical processes, however it was not definitive. The historic rain/snow/ice event in mid-February 2021 appeared to impact mass-based reductions for all metals. Qualitatively, precipitation seemed to have the largest effect on nutrient dynamics, specifically ammonia-nitrogen.

Determining exact mechanisms of pollutant removals was outside the scope of this study. An improved flow record, more targeted storm sampling, or more comprehensive nutrient profiles could aid in answering important questions on dominant mechanisms of this new constructed wetland. This study is useful in establishing a framework and baseline for understanding this one-of-a-kind regional stormwater treatment project and pursuing further questions in the future.

Acknowledgements

I want to begin by thanking Dr. Yangdong Pan and my graduate committee for allowing me the space to ask critical questions about an important project that protects water quality and provided me an unbelievable outdoor classroom for the past 3 years. I also need to thank Dr. Pan for taking that gamble by accepting a mid-career former chemistry student hoping to expand into this deeply meaningful field of environmental science. Thanks also goes to Dr. Gene Foster for encouraging me to pursue this project and providing professional support and great conversation.

A cheers also goes to members who came before me and those still here in the constantly educational and supportive Pan lab group. I appreciate you all for tolerating the method papers and differential equations I'd suggest we read, but more importantly, for your critical insights that helped me finish this project.

I also owe a lot of gratitude to my community partner and employer, Clackamas Water Environment Services, and specifically Ron Wierenga, who gave me the space and flexibility to pursue this study while working, with the goals of professional development and to better serve the County in its mission as surface water managers. I also owe a thanks to my immediate co-workers at Clackamas WES, Jim, Matt, Eric, Patrick, Melissa, Jamie, and Ron M., who supported me in my 3 years of divided attention, provided moral support during this journey, and were absolutely critical to the water quality data at the foundation of this study.

To my first family, Mom, you know I love you. Thanks for giving me a brain I can do some useful things with. Your example of compassion has been the compass I try to approach all things in life with. To my second family, Anne, Ian, John, Alexandra, and others – thanks for asking me about school, even when you knew how long I would ramble on about it. Last, but not least, to Christina. You inspired me every day to stay positive and finish what I started, especially when things got hard. I'm so excited to start this next chapter of our lives together.

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Introduction

Pollution runoff from stormwater originating in urban and industrial land areas adversely affects water quality in coastal waters (Ahn et al., 2005), groundwater (Whittemore, 2012), and surface waters (Cockerill et al. 2017, Mallin et al, 2009). In areas covered with dominant land uses such as impervious industrial, semi-industrial, or commercial land uses, runoff mechanisms typically involve dry or wet deposition of pollutants onto these surfaces and subsequent transport via precipitation events to conveyance systems (e.g., to municipal separate storm sewer systems (MS4)). In many areas of the United States, these conveyance systems transport polluted runoff efficiently and untreated to local tributaries, creeks, and streams. Typical pollutants found in runoff from urban areas include fertilizers (e.g., Nitrate-Nitrogen or Phosphorous), pesticides, heavy metals, bulk solids, thermal pollution and salts (Cockerill et al. 2017). The mechanisms of transport for these contaminants are as varied as their loading patterns but driven by environmental factors such as catchment basin size, land-use, and precipitation patterns (Ghane et al. 2016, [cite others]). The effects these pollutants have, however, are consistent and nearly universally adverse to maintaining resilient and sustainable ecosystems (NRC, 2008), with the degree of urbanization being a well-studied factor driving impacts across multiple spatial scales (Wang, et al 2001). These impacts from urban stormwater runoff demand a management response that not only addresses the pollutants in question through reduction, but can be adaptive to future growth and stressors while also providing co-benefits and ecosystem services to the surrounding human and animal communities. One commonly employed response in municipal stormwater management settings is groundwater infiltration via underground injection control devices (Bonneau et al., 2017). Another approach is a type of measure from the world of green infrastructure, with roots in domestic wastewater treatment: constructed wetlands (CW).

Constructed wetlands are resilient human-made (i.e., engineered) wetlands that employ the physical, biological, and geochemical processes that occur in natural wetlands to "treat" pollutants (Mangangka et al., 2016). They provide a wide range of provisioning, regulating, and cultural ecosystem services (Moore & Hunt, 2012; Stefanakis, 2019). An example of a provisioning ecosystem service might be food resources or habitat for fish and amphibians. A regulating ecosystem service could be reducing stormwater pollutants, mitigate greenhouse gas emissions, or attenuate peak stormwater runoff. One can envision a spectrum (Figure 1) where on one end are "gray" stormwater control measures (SCMs) such as pipes and pumps and on the other end are "green" SCMs such as stormwater ponds or CWs. Stormwater management infrastructure on the green end of this spectrum is also known as green stormwater infrastructure (GSI).

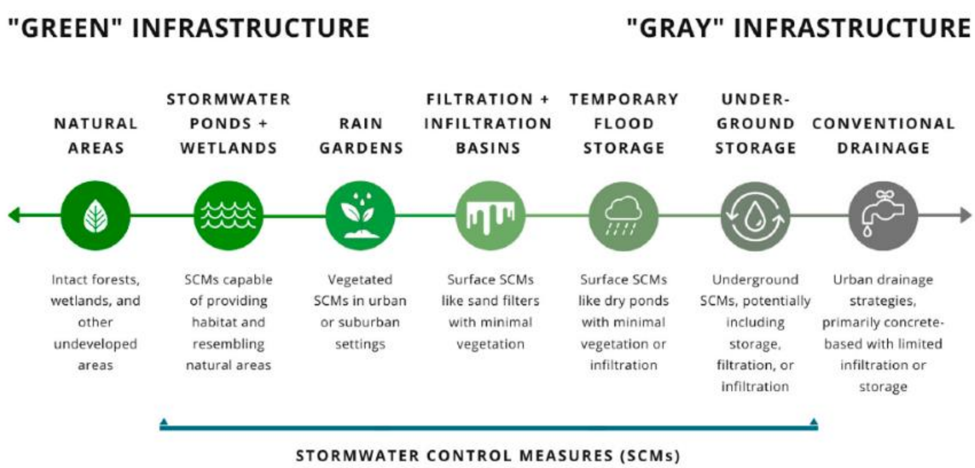

Figure 1. The Gray-Green infrastructure spectrum. (Taguchi et al. 2020)

This "gray-green infrastructure continuum" serves as a useful analytical model when considering different approaches to treating stormwater in terms of several factors important to stormwater management professionals. Bell et al (2019) explored these different factors in a decision-making analysis considering just such a continuum. Drainage area, hydrologic and water quality processes, environmental life cycle analysis, and community co-benefits of different SCMs were reviewed in their relation to a ranked list of decision factors an expert panel deemed important (Figure 2). While the authors admit their focus omitted common implementation barriers, and that their main focus was to develop decision-making support tools, this study reflects the level of attention this continuum is gaining in the stormwater management industry. The authors somewhat unsurprising finding that most projects are implemented to meet MS4 compliance goals speaks to the need of better understanding what and how green infrastructure like CWs can fit into a toolbox the regulated community can draw from. Urban stormwater pollution management is a unique problem in that the pollution sources themselves are, while well studied (McGrane, 2016), transitional, and evolving, are in some cases now historical reference points (Müeller et al., 2020).

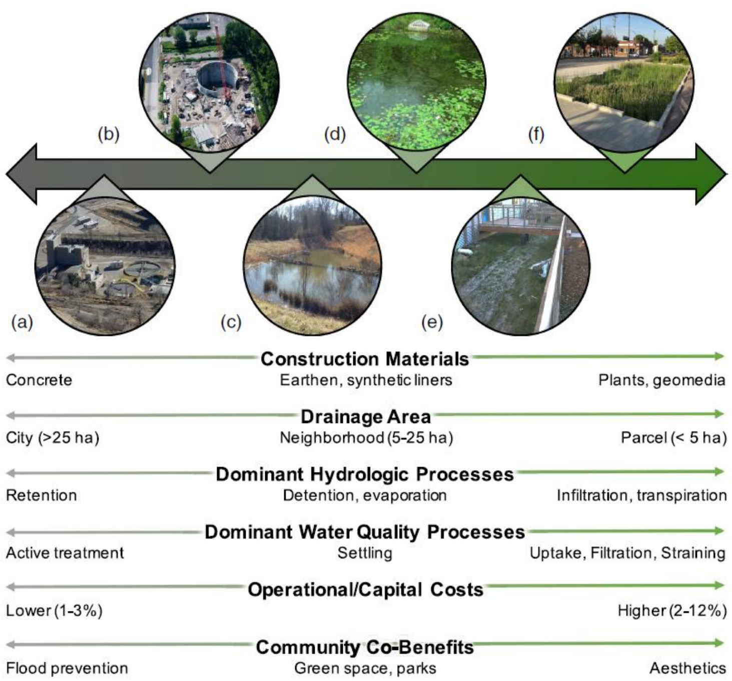

Figure 2. Perceived factors governing SCMs choice on the green-gray infrastructure continuum.

Implementation of urban GSI and other green practices are gaining widespread endorsement over typical gray infrastructure stormwater management practices because of their multiple co-benefits, resiliency to climate change, acceptance by stormwater managers particularly in the Pacific Northwest, and underlying social connections they facilitate due to their proximity to people (Shandas, et al., 2020). That said, urban GSI and CWs are not without their critiques, some of which focus on their water quality improvement shortcomings and potential for biomagnification of certain stormwater pollutants (Helfield & Diamond, 1996). Nonetheless, existing studies of CW applied to treating urban stormwater lack a complete understanding of mechanistic processes occurring within them that could help refine design criteria of these GSI (Lucas, et al., 2015). Further site-specific monitoring would also serve to address uncertainty in new CW projects and the treatment elements employed (Liu et al., 2014) and could be incorporated as part of an adaptive management approach to stormwater management as CW projects age. This study aims to ask whether a CW brought online in Clackamas County, OR in 2018 is reducing pollutants it receives from a heavily industrial urban catchment. The study also aims to qualitatively explore certain environmental variables in order to understand temporal differences in pollutant concentrations and mass reductions within the wetland and an adjacent stream, Carli Creek. These objectives can be summarized in the following questions:

- How well does the wetland reduce pollutants on a concentration- and mass-basis?
- Do weather-related or CW-specific variables explain varying treatment effectiveness?

Project Background

What are Constructed Wetlands and how do they function?

Like naturally formed wetlands, constructed wetlands function by a complex set of processes and principles that occur among the soil, organisms, and vegetation (Greenway, 2010). An example of one function that transforms pollutants in wetlands is the nitrification/ denitrification process, an inter-connected cycle between redox reactions between microbes, hydric soils, and macrophytes (Ji et al., 2020), transport dynamics caused by varying hydroperiods, and biogeochemical transformations within plant rhizospheres. A conceptual model of bacterial removal (Figure 3) highlights the many complex, interrelated factors in waterbodies that control micro-organism fate (International Stormwater BMP database [ISWBMPdb], 2022).

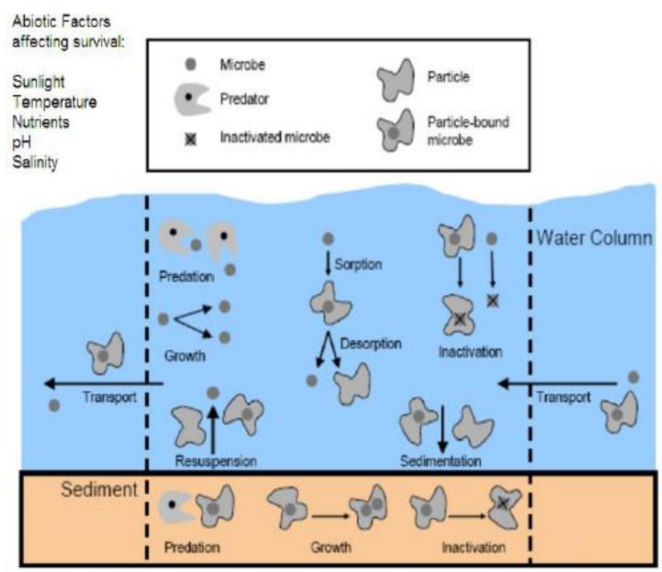

Figure 3. Factors controlling microorganism fate in waterbodies.

Also like naturally formed wetlands, constructed wetlands are susceptible to the same natural processes that shape and control their ecological functions. While the literature is rich with models and studies on CW maturation processes (e.g., allogenic versus autogenic succession theories) and collective treatment performance, there still exists a need to understand mechanistically how different individual CW components perform in removing pollutants (Malyan et al., 2021). Particularly as construction and maintenance of these projects by private and public owners can be expensive, logistically complicated, and subject to the similar constraints as gray infrastructure systems (e.g., performance breakdown, cost, design).

How are Constructed Wetlands Used in Stormwater Management?

Constructed wetlands have been used since humans began understanding natural wetland structure and function. Most CWs are unique in the type of energy required compared to their conventional treatment counterparts. Rather than requiring energy in the form electricity, manpower, and fossil fuels to operate, they rely on renewable sources such as precipitation, microbes, biomass, solar radiation, soils, and wind (Knox et al., 2010). The trade-off with respect to resource intensity however is that constructed wetlands often require much more land and much more time (i.e., residence time) to efficiently treat pollution. Nonetheless they can be found in real-world applications globally treating a variety of waste streams such as groundwater, municipal sanitary wastewater, industrial wastewater, and stormwater. They've also proved effective for removal of most types of pollutants one might assume advanced, expensive conventional treatment works would be required for, particularly in urban environments. These include nutrients, solids, metals, pathogens, and in some cases man-made synthetic chemicals (Nguyen et al., 2021, Walaszek et al., 2018a, Walaszek et al., 2018b). While the basic role these features play is water quality improvement (Shutes, 2001), the way they're designed, used, and performed is the focus of this discussion. It is worthwhile to first discuss and understand use of CWs in wastewater treatment, as that use is better understood by the value that effective water pollution control is given in a regulatory sense. Furthermore, the literature of domestic wastewater treatment via CWs is richer than stormwater treatment.

Since the first drop of wastewater was spilled into a wetland, they have been effectively treating human-derived, “domestic” wastewater. History has transformed these natural wetlands into the hybrids of today termed constructed wetlands. Today, they’re used to treat wastewaters varying from domestic wastewater, acid-mine drainage, oil-refinery wastewater, cooling tower recirculation water, landfill leachate, and agricultural runoff across the globe (Babatunde et al., 2008, Kadlec et al., 2000, Pat-Espadas et al., 2018). The specific suite of pollutants each of these wastewaters convey are unique to the processes occurring upstream. This requires careful planning of various CW design elements such as size, hydraulic dynamics, storage capacity, slope, vegetation, and maintenance practices. To add even further design consideration, many CW vary in the precise transportation process of the fluid through the system. For example, the surface-flow types typically have inflow and outflow locations above the ground surface, mimicking a regularly flooded wetland morphology. Sub-surface flow CWs have an inflow above the surface of the sediment and either an outflow below the surface or lack one entirely. Other common flow designs include vertical flow and hybrid styles employing different cells in series or parallel with each other. All of these flow design wetlands, to one degree or another, interact with the groundwater table (unless completely lined) and therefore are all affected to some degree by groundwater and hyporheic processes. Wastewater treatment by constructed wetland has been proven to work, and the same processes that remove nutrients, solids, pathogens, and metals in those waste streams will remove them in stormwater runoff.

Constructed Wetland Treatment Effectiveness

Much like wastewater treatment, unit elements of constructed wetland’s and their treatment effectiveness on diffuse sources like urban stormwater runoff makes them valuable tools as sustainable, low-cost environmental management options for stormwater (Shutes, 2001, Schulz & Peall, 2001, Guittonny-Philippe et al., 2014). Constructed wetlands can be categorized by many factors, including their size, catchment size, age, major vegetation types at initial planting, major pollutants removed, and pollutant removal efficiencies. CWs are also categorized in the flow regime they're designed to experience, as these regimes affect the hydraulic efficiency of CWs, their ability to utilize the full area, volume, and abundance of planted vegetation (Persson et al., 1999). These regimes range from surface-flow ("SF"), horizontal sub-surface flow ("HSSF"), and vertical sub-surface flow ("VSSF") (Greenway, 2004) and reflect the dominant hydrologic flow paths for stormwater treatment CWs. An overview of recent studies in stormwater treatment constructed wetlands is given below (Table 1). It is worth mentioning that mecocosm experiments and CWs receiving other types of inflow (e.g., groundwater) experiments were excluded in this table, even though the literature is rich with valuable data and lessons-learned that are applicable to stormwater CWs (Nilsson et al., 2020, Li et al., 2019, Payne et al., 2014).

Table 1. Summary of stormwater CW water quality treatment effectiveness field studies (microcosm studies excluded).

	Location	Major treatment units[1]	Flow Regime	CW Size, ha	Catchment Size, ha	Major vegetation types	Age, yrs	Major Pollutants studied[2]	Removal Efficiency[3] Conc.	Mass	Reference(s)
1	Strasborg, France	SP→CW	VSSF	0.009	2.71	*Phragmites australis*	6	T/D HMs, PAHs	0-96 50-92	94–100 66–100	Walaszek et al., 2018
2	Canning, Australia	RP→CW→ RP→CW→ RP	In series SF and HSSF	1	129	*Beumea* spp., *C. appressa*, *J. kraussii*,	0-6	T/D N & P, T/D C	-58 – 63 x̄ = 66	-1–76 10–99	Adyel et al., 2016
3	Växjö, Sweden	SP→CW	SF	6.8	320	Unidentified	3, 9, and 16	HMs, N, P, Solids	76–97 52–59 84–89 92–96	42–96 41–68 65–92 49–97	Al-Rubaei et al., 2016 & Semadeni-Davies, 2006
4	Sydney, Australia	SP→CW	SF	0.08	48	"Reeds"	1	HMs[4], N, P, Bacteria	-5–89 -34–70 -14–39 26–99	NR	Birch et al., 2004
5	Virginia, USA	CW	SF	0.07	1.3	Unidentified	3	HMs, N, P, Solids	-22–50 -27–69 20–35 m=58	-69–42 -64–63 -12–0 m=50	Carleton et al., 2004
6	Windsor, Australia	CW	SF	0.45	75	"Emergent indigenous macrophytes"	2	Bacteria	79–87	NR	Davies & Bavor, 2000
7	North Carolina, USA	CW	SF	0.14	46.5	*N. odorata*, *P. cordata*, *S. cernuus*, *P. virginica*, *J. effuses*	5	N, P, Solids	-8–41 8–30 x̄=15	7–27 14–36 x̄=-8	Merriman & Hunt, 2014, Lenhart & Hunt, 2011
8a	Texas, USA	Channel→ CW	SF	12.1	2060	*S. californicus* *S. americanus*	Unk.	N, P, Solids *E. coli*	-3–11 m=33 m=49 m=-8	NR	Guerrero et al., 2020
8b	Texas, USA	Channel→ RP→RP→ CW	SF	11.3	500	*S. americanus*	Unk.	N, P, Solids *E. coli*	2–31 m=17 m=56 m=36	NR	Guerrero et al., 2020
9	California, USA	CW	SF	23	0.637	*Typha* spp., *Scirpus* spp., *Juncus* spp., *Carex* spp., *Lemna* spp.	5	N, P, Solids	24-83 59-70 m=74	59-76 71-79 88	Hayvaert et al., 2006

[1] – SP = Sedimentation Pond, RP = Surface-Flow Retention Pond
[2] – T = Total, D = Dissolved, HMs = Heavy Metals (e.g., copper, lead, zinc, etc.), PAHs = Polycyclic Aromatic Hydrocarbons, N = Nitrogen (i.e., Total N, TKN, or NO_x), P = Phosphorous (i.e., Total P or ortho-phosphorous), C = Carbon, OCs = Organochlorine Pesticides, PCB = Polychlorinated Biphenyl compounds, Bacteria = Indicator organisms (e.g., fecal coliform, *enterococcus*, or *E. coli*), O & G = Oil and Grease
[3] – When available, metric is for entire system. Given as a range (minimum – maximum), arithmetic mean ($\bar{x}$), geometric mean ($g\bar{x}$) or median (m). NR = Not reported
[4] – This study saw significant export of iron and manganese, up to 269% and 477% respectively, for one event.

Another regularly updated, publicly-accessible source of green infrastructure effectiveness is the International Stormwater BMP database ("ISWBMPdb", 2022), a database comprised of over 700 studies, that provides graphical and tabular summaries for 12 BMP types and 20 pollutants (Table 2). These data were used in the design of the Project and are useful in comparing future concentrations at the site.

Table 2. 2020 International Stormwater BMP database BMP and pollutant categories.

BMP	Pollutant Category		
Detention Basin	Solids	Total Suspended Solids (TSS)	
Retention Pond		Total Dissolved Solids (TDS)	
Wetland Basin	Bacteria	Fecal coliform	
Wetland Channel		*E. coli*	
Grass Swale		*Enterococcus*	
Grass Strip	Metals†	Arsenic	Iron
Bioretention		Cadmium	Lead
Media Filter		Chromium	Nickel
High-rate Biofiltration		Copper	Zinc
High-rate Media Filtration	Nutrients	Total Phosphorous	
Hydrodynamic Separation Devices		Orthophosphate	
Oil/Grit Separators and Baffles		Dissolved Phosphorous	
Permeable Friction Course		Total Nitrogen	
Porous Pavement		Total Kjeldahl Nitrogen	
		Nitrate and Nitrate+Nitrite (NOx)	
		Ammonia as N	

† = Total and Dissolved

While there are significant challenges in identifying specific pollution profiles at the catchment-scale, stormwater control measures and green infrastructure are widely adopted and considered generally effective at mitigating the effects of these surfaces on downstream aquatic resources. Constructed wetlands, as man-made analogues to natural wetlands, offer many benefits to assimilating and treating urban stormwater pollution. Benefits such as the aesthetic appeal of "natural" wetland landscapes over gray stormwater infrastructure may appear obvious, but the physicochemical functioning of CW are difficult to quantify. To name just a few factors affecting stormwater treatment performance in the literature: flow dynamics (Wadzuk et al., 2010; Feng et al, 2014), sediment composition and porosity, vegetation planting and management (Zhu et al., 2017), incoming pollutant profile (Chen and Chang, 2014; Knox et al., 2006), the collection of interrelated design features constructed on-site (e.g. microtopography, storage capacity, slope, aspect, etc.), and landscape/weather factors (e.g. antecedent dry period, rainfall, land-use). Wu et al (2016) reviewed indicator pathogen (e.g., *E. coli*) removal in constructed wetlands and identified multiple physicochemical and biological processes including predation, sedimentation, adsorption, and vegetation presence. Heavy toxic metals and their removal efficiencies and mechanisms are also frequently studied (e.g., arsenic in Lizama et al., 2011) in the context of stormwater CWs.

Evaluating Effectiveness

Evaluating stormwater SCMs has been a goal of ecological engineering since stormwater runoff was identified as one of the largest sources of water pollution in the United States (NRC , 2008). Many approaches have been taken, attempting to strike a balance between a true accounting of SCM performance and efficient use of monitoring resources (e.g., laboratory analysis, flow measurement, etc.). A true accounting of SCM performance is important for reasons beyond the value of accurate field data – SCM performance data inform engineers', planners', and resource managers' decisions on important choices they make with limited budgets. For example, SCM performance data aids in choosing what works best at a particular site subject to particular stormwater runoff quality to achieve desired effluent characteristics. Performance data can also inform best management practices (BMPs) for maintaining constructed wetlands long-term. In the real world of project budgets, stringent water quality standards, and increasing urbanization, robust, accurate SCM/BMP performance data is as good as gold. A common approach in evaluating any stormwater SCM's effectiveness is through pollutant removal efficiencies.

Removal efficiencies treat stormwater SCMs like constructed wetlands as “black boxes”, or systems so complex that the most straightforward approach to understanding them is by measuring what goes in and what comes out. This performance metric, while elegant in its simplicity and useful in some investigations, lacks the sort of detail to inform resource managers who employ these SCMs about anything other than does the pollutant that goes into the BMP increase or decrease when it come out. In systems with simple design features or single-stage treatment processes (e.g., catch basins under roadside curbs), this might be sufficient information. In more complex treatment systems such as those mimicking natural treatment processes (e.g., CWs), more interrelated processes occur simultaneously, confounding conclusions and obscuring the mechanism and drivers of high performance (at least in the sense of water quality improvement). This type of metric may also be insufficient to predict how management actions might degrade or improve SCM/BMP effectiveness. One more shortcoming of this metric is it neglects the landscape in which the process is occurring (i.e., the metric doesn’t account for flow, precipitation, surrounding land-use effects, and often, temporal variability like seasonality). Lenhart and Hunt (2010) studied how removal efficiency compares as a metric against three others at the South Carolina constructed wetland, River Bend. The three other metrics included % removal by loads (flow x concentration), influent/effluent concentrations in the context of ambient stream concentrations, and influent/effluent concentrations in the context of other storm-water related studies in North Carolina. Their evaluation concluded that depending on the metric used, the performance of the CW varied from poor (removal efficiency), to mixed (ambient water quality), to well (loadings). Their study suggests accounting for landscape and environmental factors is prudent in assessing constructed wetland performance.

Detailed descriptions of researchers conducting constructed wetland performance studies illustrate the diversity in study design and conclusions drawn. In southern Sweden, a surface-flow 6.8 ha CW receiving runoff from a 320-ha catchment (2.1% by area) was examined 19 years after construction to assess maintenance effects on performance (Al-Rubaei et al., 2016). Load (or mass of pollutants) removal efficiency metrics were evaluated to show that performance remained high with minimal maintenance occurring in the part of the CW designed as a detention pond. In Pennsylvania, USA, researchers from Villanova University studied a surface flow 0.4 ha CW treating an 18.2 ha catchment's runoff (2.2% by area) to examine effects of incoming peak and base flow conditions on the performance of solids, nutrients, metals, and *E. coli* treatment (Wadzuk et al., 2010). They accounted for wetland maturity in their design by sampling over two different intervals (i.e., 4 and 8 years after construction) while active vegetation management was occurring. Their study concluded that, while the CW improved effluent water quality under all seasons and flow conditions, storm flows had lower inlet concentrations and increased retention times had multiple benefits (no metal leaching, solids settling, and attenuation of peak flow impacts). Researchers in Strasbourg, France evaluated a hybrid surface-flow/vertical-subsurface flow CW (28 m^3 pond before a 0.01 ha "filter) treating a 2.7 ha catchment (Walaszek et al., 2018b). Their study focused on micropollutant (PAHs, metals) treatment in wet and dry events of this 6-year-old CW with the goal of recommending management actions. Assessing removal efficiency by concentration and load (i.e., mass) again showed generally high values, although resuspension and output of particulate zinc was common. Their analysis of pond sediments showed a potential for re-use as road backfill, an alternative to landfilling (Walker, et al.). These types of studies illustrate that given enough maintenance, even highly stressed (i.e., high catchment-to-constructed wetland surface-area ratios) CWs remain effective at improving water quality long after they're built.

Community Partner and Site Information

Clackamas Water Environment Services (CWES) is a department in Clackamas County responsible for managing the MS4, its natural areas, and the stormwater infrastructure which treats stormwater runoff. CWES is the community partner for this project and was the employer of the author through the duration of the project. As a member of the "Clackamas County Group," CWES administers, facilities, and coordinates activities among the 12 individual co-permittees of the group and possesses one of only eight Phase I MS4 permits in Oregon. Phase I MS4 permits are issued to entities with total populations greater than 100,000. MS4 permits more generally are members of the family of National Pollutant Discharge Elimination System (NPDES), a system derived to control pollution whose authority is codified in the federal Clean Water Act (33 U.S.C. § 1251*et seq.*) and its subsequent amendments. These permits are developed and issued by the Oregon Department of Environmental Quality (DEQ), Oregon's control authority for such permits. Legal authority to issue such permits is granted in Oregon Revised Statute (ORS) 468B.050. The current permit (No: 101348) was issued in 2021 and will expire in 2026. To comply with this permit, Clackamas County Group members can collectively or individually develop Stormwater Management Plans which detail implementation steps for complying with the issued permit's conditions. Examples of such details include dry-weather outfall monitoring, in-stream and stormwater monitoring, BMP activities and construction projects, and evaluation techniques of these efforts to guide implementation priorities.

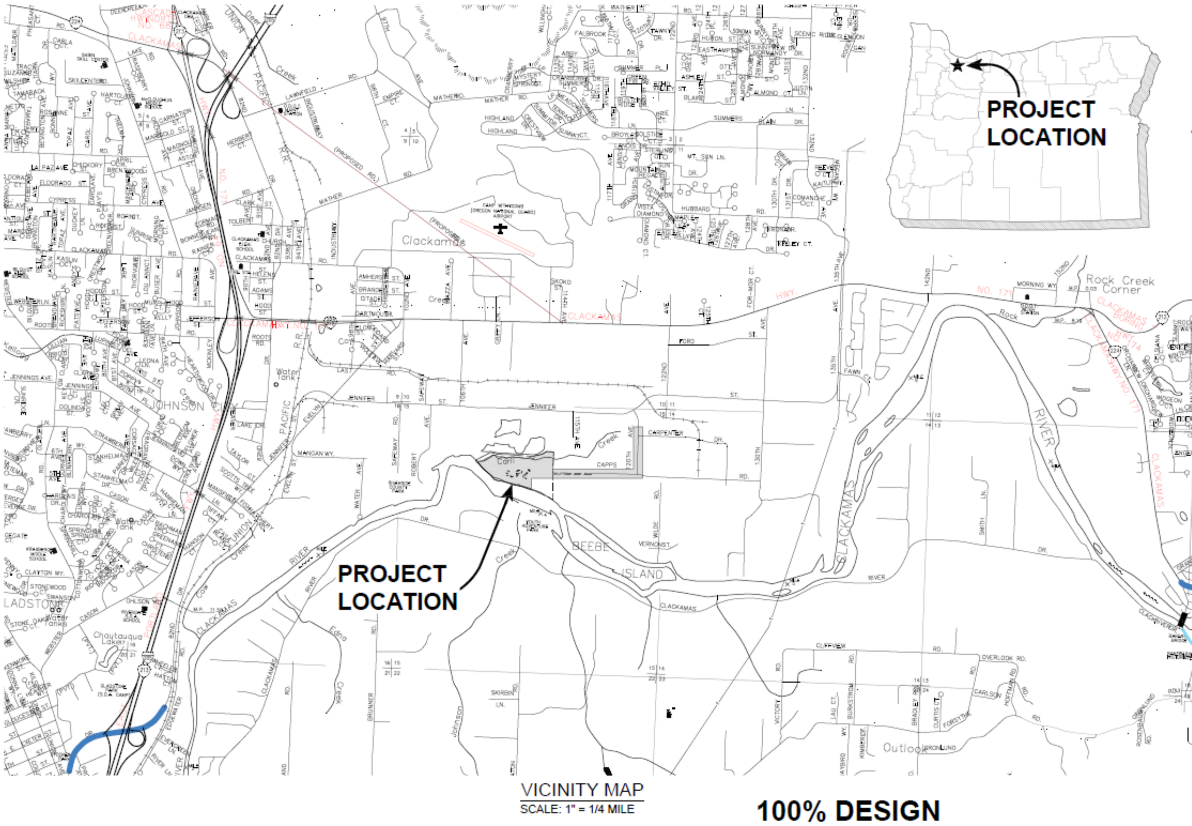

Figure 4. Global context map of the Carli Creek Water Quality Map.

While this study's primary focus is understanding constructed wetland performance and potential weather-related variables which could explain variance in that performance, several regulatory criteria, indices, and benchmark levels are published in Oregon for assessing pollutant concentrations in rivers and streams. A list of these criteria is described below as reference and will be evaluated against the collected data to place the concentrations measured in context.

Common in-stream criteria used in Oregon typically fall under the umbrellas of the Oregon Water Quality Standards, permit-based benchmarks, or Total Maximum Daily Load (TMDL)-related loading calculations. Challenges with using these are typically related to insufficient criteria for pollutants or inappropriate applicability. Oregon's Water Quality Standards (OAR 340-041-0001) are a set of scientifically developed, publicly-reviewed benchmarks that help resource managers assess if the water quality of a particular body of water meets its designated uses. Examples of the types of pollutants these Standards set benchmarks for are copper, lead, zinc, temperature, PCBs, dissolved oxygen, aluminum and a variety of narrative-based criteria. A couple of examples of aesthetic narrative criteria include “the formation of appreciable bottom or sludge deposits or the formation of any organic or inorganic deposits deleterious to fish or other aquatic life or injurious to public health, recreation, or industry may not be allowed” (OAR 340-041-0007(11)) and “aesthetic conditions offensive to the human senses of sight, taste, smell, or touch may not be allowed” (OAR 340-041-0007(13)).

Oregon’s 1200-Z Permit (Oregon NPDES 1200-Z General Permit, effective August 1, 2017) controls industrial discharges of stormwater that may reach public water ways, either directly or indirectly through conveyance systems. The applicability of this permit is specific to the sources listed in it and isn’t generally used to assess stormwater from municipal systems. However, its discharge benchmarks could provide useful reference concentrations (e.g., for Total Copper, Lead, and Zinc, pH, TSS, Total Oil and Grease, and *E. coli*) for assessing stormwater pollution from industrialized areas such as the Carli Creek catchment.

A last framework for contextualizing water quality criteria are TMDLs. These are basin-, pollutant-, load-specific written plans and analyses that establish and ensure that waterbodies will attain and maintain water quality standards. Attainment would, for example, be a TMDL goal to return a waterbody back to supporting the most sensitive beneficial uses, while maintaining is a built-in goal of TMDLs to which accounts for uncertainty in order to maintain the attainment status. Examples of effective and proposed TMDLs in Oregon are the Phosphorous (e.g., 0.14 mg/L P dry-season summer median below Dairy Creek) and Mercury TMDLs in the Tualatin subbasin and Willamette basins, respectively. Data generated at CWES in the scope of MS4 compliance monitoring has been assessed against these three umbrellas of standards historically to answer important questions about its pollution prevention efforts.

Land-uses which collect and convey stormwater to the rivers and streams in CWES' service areas are widely variable. There are concentrated industrial areas, mixed-use commercial/residential areas, and subdivisions of detached single-family homes. Within CWES, the Environmental Services division is, among other things, tasked with protecting water quality by reducing pollution in rivers, streams, and wetlands caused by stormwater runoff. In partial fulfillment of the objective of assessing pollution from varying land-uses, a long-term (1994-present) monitoring site has existed at a piped section of Carli Creek, before it daylights. Analysis of the historical monitoring data indicated water quality exceedances for copper, zinc, and *E. coli* bacteria. To help ameliorate these impacts from the upstream runoff, CWES engaged a small agricultural landowner who had earned a living for several generations on the banks of Carli Creek. What was the purpose of this engagement? To build support and acquire the land to construct a multi-benefit regional water quality improvement project.

Approximately eight years ago, CWES acquired a 6-ha parcel of agricultural land along the banks of the Clackamas River, near the mouth of Carli Creek. Carli Creek receives stormwater runoff from a highly industrialized area (~162 ha) in Clackamas County and prior geomorphic, macroinvertebrate, and water quality monitoring has indicated the creek's ecological functions are degraded (Waterways, 2018). These functions included providing habitat for local birds and plants, benthic macroinvertebrates, and water quantity and quality pollutant reduction from stormwater runoff delivered to it. The ecological status of Carli Creek is a prime example of the urban stream syndrome. The urban stream syndrome can conceptually be described as a suite of symptoms common to streams which receive drainage and runoff from urbanized lands. Common symptoms include "a flashier hydrograph, elevated concentrations of nutrients and contaminants, altered channel morphology and stability, and reduced biotic richness, with increased dominance of tolerant species" (Walsh, 2005, p 707).

Acquisition of this adjacent property provided an opportunity for CWES to construct a $3.5 million-dollar innovative natural constructed wetland, named the Carli Creek Water Quality Project (CCWQP or "Project"). One of the five design goals stated that "target pollutants [would be] reduced" describing that the project would perform because the "design features [a] treatment train with processes that are effective for removal of target pollutants." (Herrera, 2015a). Estimated reductions were derived from the International Stormwater BMP database, a routinely updated clearinghouse of treatment information on stormwater for various stormwater treatment systems. While the Project is unique enough to have no direct corollary in the database, estimated effluent concentrations for swales and wetlands are on average 21.6 and 9.4 mg/L total suspended solids, 5.6 and 2.5 µg/L dissolved copper, and 19.8 and 7.6 µg/L dissolved zinc, respectively (Herrara, 2015b). More recent performance summaries for a variety of BMP categories relevant to the CCWQP are shown in Table 3 below (WRF, 2020).

Table 3. Median effluent concentrations of select parameters for 3 stormwater BMPs.

Parameter	Fraction	Units	BMP Category, median (95% CI) effluent concentrations: Wetland Basin	Retention Pond	Bioretention
TSS		mg/L	14.0 (11.5, 15.2)	12.0 (11.0, 13.0)	10.0 (8.0, 11.0)
TDS		mg/L	149 (92.0, 168)	178 (152, 206)	210 (175, 298)
E. coli		MPN/ 100 mL	884 (311, 1320)	708 (156, 1370)	158 (46.5, 212)
Total Phosphorous		mg P/L	0.122 (0.108, 0.133)	0.120 (0.104, 0.129)	0.240 (0.190, 0.270)
Ammonia Nitrogen		mg N/L	0.0600 (0.0473, 0.0608)	0.0785 (0.0670, 0.0901)	0.0500 (0.0500, 0.0600)
Nitrate & Nitrite N (NO_x)		mg N/L	0.234 (0.170, 0.312)	0.163 (0.140, 0.190)	0.441 (0.380, 0.507)
Cadmium	Total	ug/L	0.170 (0.114, 0.200)	0.200 (0.154, 0.200)	0.0825 (0.0647, 0.100)
	Dissolved	ug/L	0.300 (0.300, 0.500)	0.125 (0.125, 0.125)	0.0668 (0.0444, 0.0885)
Copper	Total	ug/L	3.32 (3.00, 4.00)	4.90 (4.42, 5.00)	7.13 (6.40, 8.20)
	Dissolved	ug/L	2.29 (1.77, 3.33)	3.50 (3.19, 3.80)	7.54 (6.50, 8.40)
Lead	Total	ug/L	1.68 (1.00, 2.00)	3.00 (2.37, 3.00)	0.932 (0.723, 1.07)
	Dissolved	ug/L	0.602 (0.370, 0.851)	0.465 (0.262, 1.00)	0.0739 (0.0506, 0.0878)
Zinc	Total	ug/L	20.1 (17.0, 23.0)	21.2 (20.0, 23.0)	12.8 (11.0, 14.0)
	Dissolved	ug/L	8.35 (6.62, 9.00)	16.0 (13.9, 17.6)	12.5 (9.00, 13.8)

Although, predicted effluent concentrations may be lower than those observed historically, no post-construction water quality monitoring was budgeted for in the Project. Further, due to flow-rerouting performed as a critical component of the project, the historic up-stream site on Carli Creek may no longer be appropriate for characterizing the pollutant loads delivered to the creek. For example, variability in pollutant concentrations is quite high based off a preliminary analysis of historic data and new conveyance structures built as part of the Project now split and re-deliver flows in a new pattern that didn’t before exist. Therefore, comparing performance under varying natural conditions will be valuable for optimizing long-term management of the Project. A released Mercury TMDL in the Willamette Basin will also have regulatory implications for stormwater management utilities (e.g., Mercury Minimization Plan conditions in NPDES permits) and mercury is known to be transported in stormwater runoff (Eckley, 2008) through multiple mechanisms. Reductions of this pollutant and others frequently detected at high concentrations is critical not just for regulatory reasons, but for providing safe beneficial uses of Carli Creek, a tributary of the Clackamas River.

Project Study Area and Surroundings

The study area is situated in Clackamas County, Oregon. East of I-205 in the northwest of the county is a sprawling area of land including rapidly developing urban areas. The project was conducted in what is considered the Clackamas Industrial Area (CIA), a large (>404 ha) district including concentrated commercial and industrial uses (Figures 5, 6, and 7, Herrera).

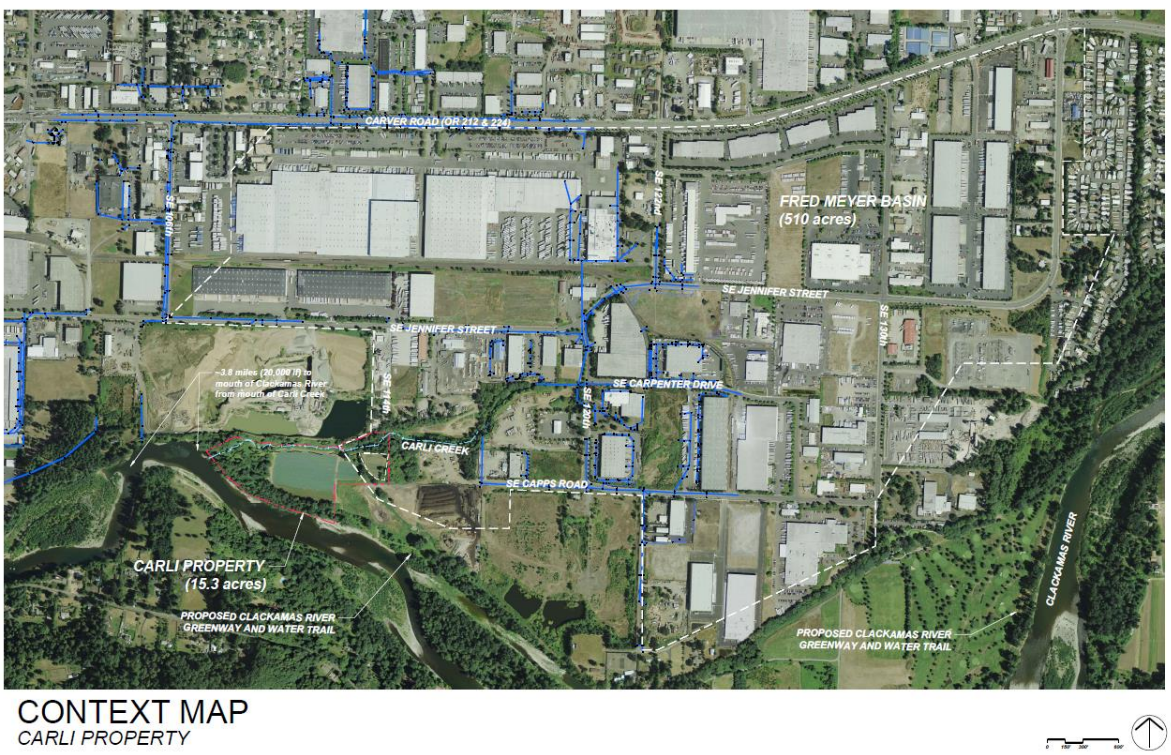

Figure 5. Map of the surrounding catchment, pre-construction (ca. 2014), with the Carli Property in the lower right corner.

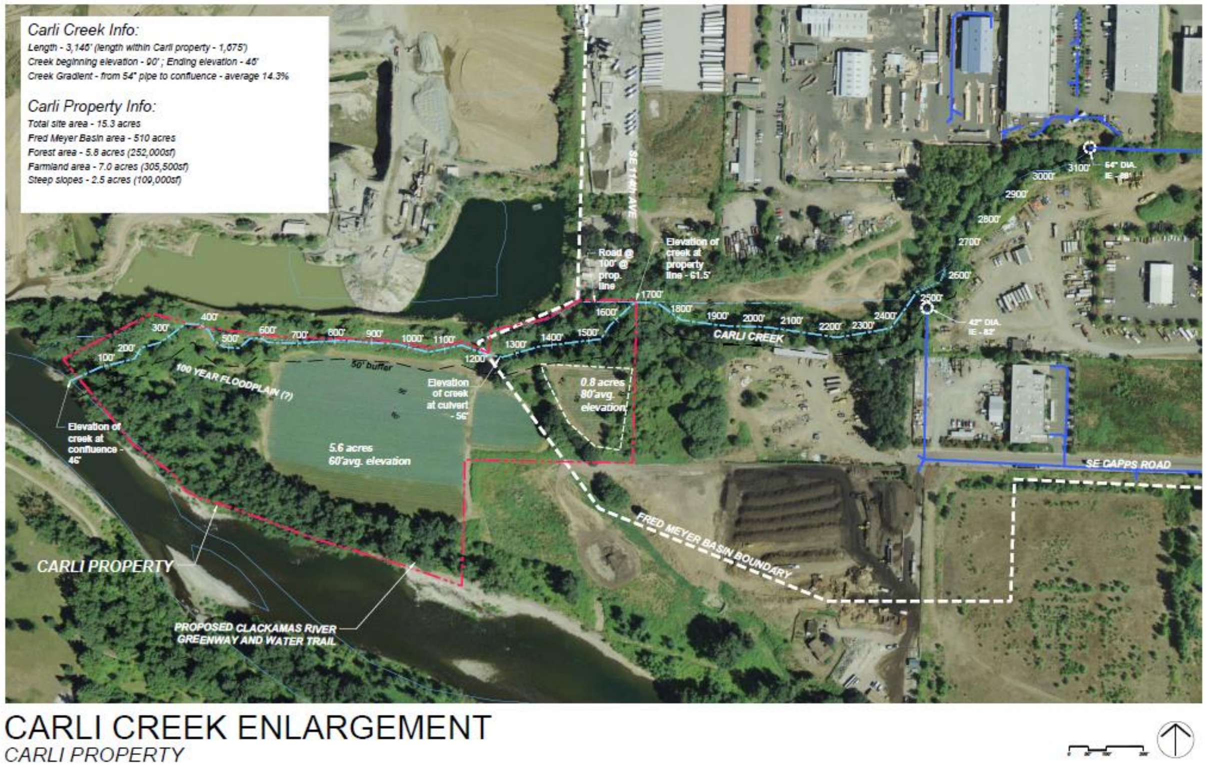

Figure 6. The Project area (red dashed line) pre-construction (ca. 2014) showing Carli Creek's stream path (light blue-green line) and relevant stormwater outfalls (white, dashed circles)

Much of the surface area is fully impervious. A land-use analysis above the historic monitoring point maintained by WES showed that the upstream catchment is 87.5% industrial (personal communication with *S. Ottersen*). Runoff from these industrial acres flow to a CW a fraction of the size (i.e., 0.73 ha of area, a 1:243 ratio to catchment area) with a total runoff storage of 4070 m^3. Businesses typical to the area include: landscape materials companies, brick manufacturers, centralized transportation hubs, military bases, municipal firehouses, road and paving, centralized waste treatment facilities, and assorted large food distributors and manufacturers. The area is built in the historic floodplain of the Clackamas River and has a typical alluvial, surficial surface deposits (Department of Geology and Mineral Industries [DOGAMI], 2022).

Project Construction

The construction portion of the CCWQP had three main objectives: restore Carli Creek's in-stream and riparian habitat, install diversion structures in the MS4 to redirect stormwater to a constructed wetland (aka, the “treatment terrace”), and maximize pollutant removal efficiency through design elements on the treatment terrace. Project design goals of these three objectives are shown in Table 3 below (adapted from Herrera). Not all of these were studied in this assessment project.

Table 4. Project design goals from the general contractor for the CCWQP.

	Goal	Performance Measure
Vegetation	Area of eradicated weeds	15 ac (6.07 ha)
	Area of emergent wetland habitat created	1.7 ac (0.69 ha)
	Area of shrub wetland habitat created	2.2 ac (0.89 ha)
	Area of riparian forest created	2.1 ac (0.85 ha)
Habitat	Total number of logs in-stream	206
	Total number of logs in floodplain	216
	Area of Carli Creek floodplain habitat reconnected	0.27 ac (0.11 ha)
	Number of large woody debris structures (e.g., beaver analog structures)	77
Water Quality	Percentage of runoff diverted to facility	74% of a 16-year rainfall event
	Percentage of runoff infiltrated at the treatment terrace	61% for average 30-day growing period 53% of a 6-month storm event 24% of a 100-year storm event
	Peak Flow reduction	20% of 6-month peak flow 10% of 100-year peak flow
	Target Pollutants (i.e., Zinc, lead, E. coli) reduced	Design features treatment train with processes that are effective for removal of target pollutants.

The relevant sections for my project will be: (1) the treatment terrace itself and its associated treatment units, and (2) Carli Creek. Weir walls were installed in the MS4 sections of Carli Creek upstream for a specific purpose but were not studied in this project. The purpose of these weir walls was to allow sufficient base flow to the creek (~0.028 m^3/s) while diverting remaining small and moderate stormwater flows towards a hydrodynamic separator device and subsequently, the treatment terrace. With the purpose of understanding how stormwater flows to Carli Creek and the CCWQP, a diagram of the diversion structures, weir walls, and subsequent outfalls is shown below (Figure 7 and 8).

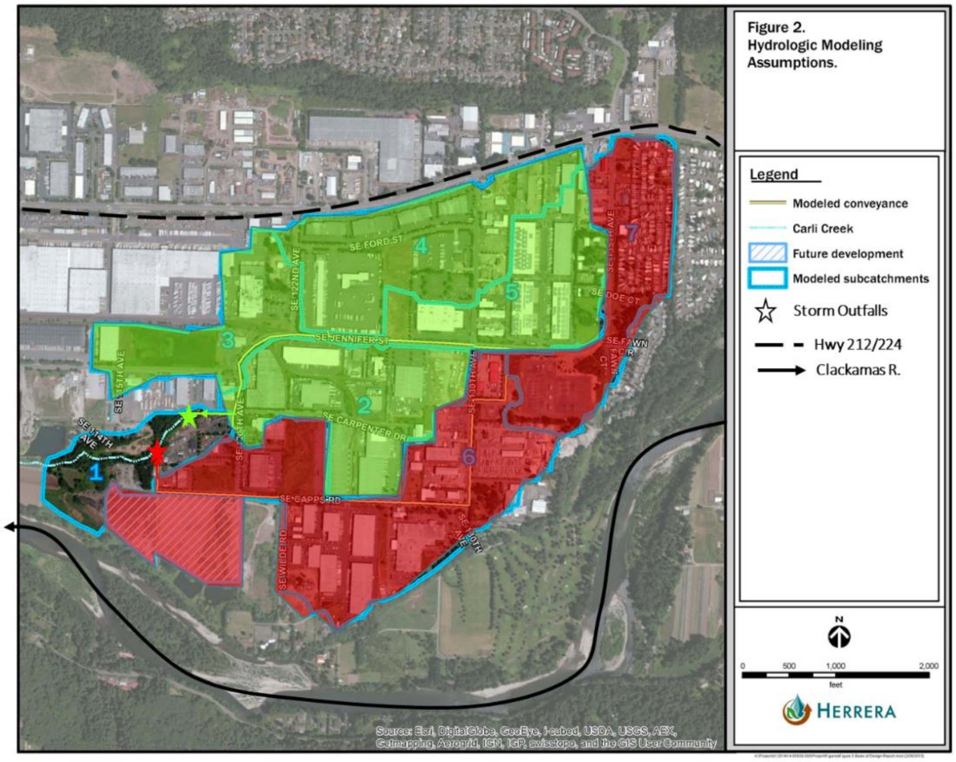

Figure 7. Upstream Sub-catchments and their MS4 outfalls, pre-construction.

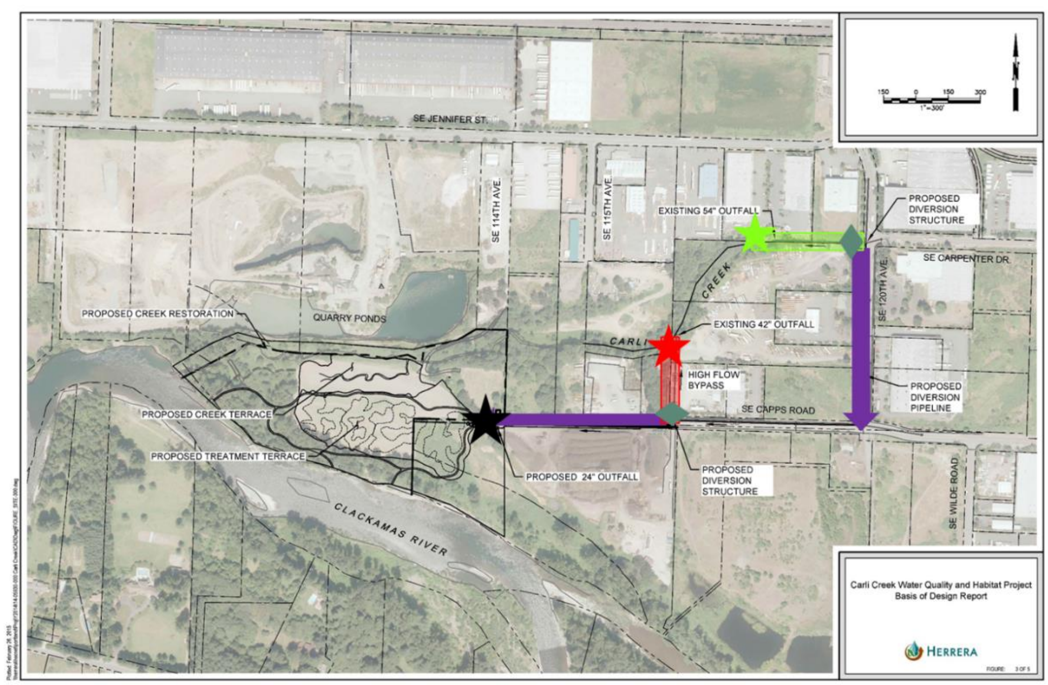

Figure 8. Schematic of installed diversion structures (diversion pipes, purple arrows; weir walls, blue-green diamonds), and new 24" MS4 outfall (black star) which discharges to the beginning of the constructed wetland treatment terrace.

The treatment terrace itself is a simple linear surface-flow constructed wetland. The flow path through the wetland is in the following order: step pools (SP), retention pond (RP), 3 bioinfiltration/treatment cells (BR) in series, consisting of the treatment terrace (TT), followed and a large backwater channel (BW) hydrologically connect to Carli Creek (Figure 9). The retention pond's overflow weir (at 65 ft elevation) adjacent to the maintenance road is designed to allow high flows to bypass the flow control structure in the pond and be routed directly to the bioretention cells. Size, vegetation, and storage characteristics of the project itself are summarized in Table 5 below. The open-channel portion of the creek is approximately 0.94 km long, initially meandering through a forested up-land reach where the large outfalls are the main hydrologic connection to the catchment. The creek continues through a low gradient reach, passing under a large corrugated metal culvert adjacent to the constructed wetland, and finally through a second moderate-gradient wooded reach before the creek joins the Clackamas River (at RM = 3.2), approximately 300 m upstream of public drinking water intakes.

Table 5. CCWQP Characteristics

Commissioning Date	Fall 2018
Catchment Size	438 ac, 177.3 ha
Catchment Land Use mix	3.1% commercial, 0.1% natural resources, 87.5 % industrial (light/general), 9.3% Residential
Retention Pond, size	0.2 ha
Pond storage volume	1789 m^3
Bioretention Cells, size	0.69 ha
High:Low Infiltration soil ratio	3.7:1
Total storage volume	2245 m^3
Total mitigation wetlands created	0.49 ha
Treatment Terrace size	0.89 ha
Backwater Channel	0.43 ha
Linear Stream restored	~1700 ft (518 m)
Large wood structures installed	77
Beaver Analog Structures installed	7
Number of Plantings	70,000+
Dominant Macrophytes	*Juncus* spp. and *Carex* spp.
Seeding mixes	2.11 ha (19% wetland, 44% riparian, and 37% oak woodland)

Figure 9. Aerial map of the treatment terrace showing separate regions of the treatment terrace ("TT"), Carli Creek, and the Clackamas River. SP=step pool, RP=retention pond, BR=bioretention cells, BW=backwater channel.

As the ongoing development in the CIA altered acres of pervious, vegetated surfaces to impervious, paved surfaces, the creek which bordered the north side of the Carli property began to degrade. Another consequence of development in the CIA was the burial and piping of the upper portions of Carli Creek under infrastructure. This eliminated roughness and morphological complexity native to natural, open-channel streams. The effects of creek burial caused characteristic "flashy" flow rates resulting from wet weather events (Baker et al., 2004), inaccessibility to the creek's former habitat (habitat reduction), and other adverse downstream effects such as alteration of natural channel forming processes which drive habitat complexity and support diverse fauna. Fortunately, extensive national studies in the USA and prior data exist above the CCWQP complex to contextualize current water quality.

A conceptual model of the pollutant transport, transformation mechanisms, and environmental factors at the Project is shown in below in Figure 10.

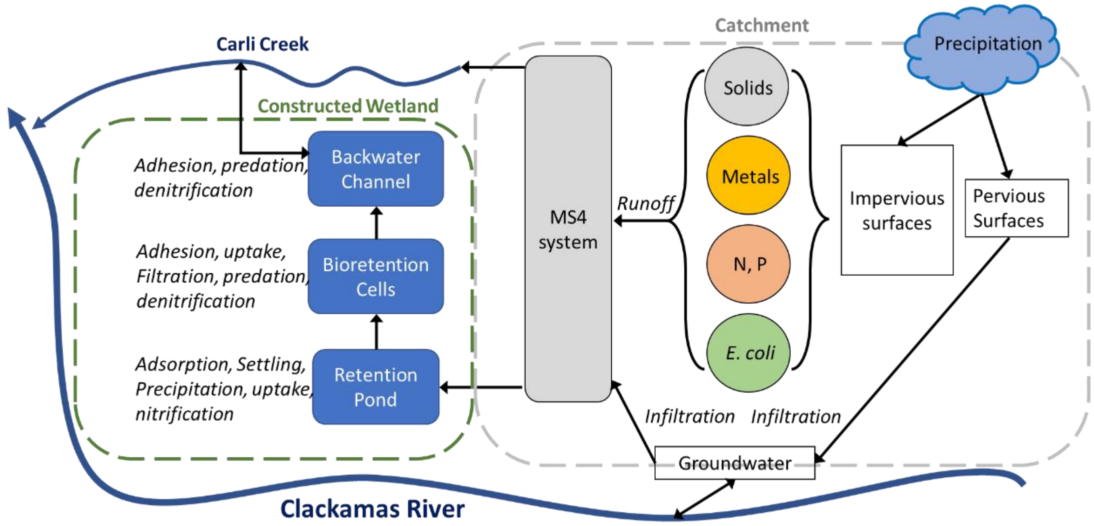

Figure 10. Conceptual Model of the Project and the associated MS4 system.

Study Objectives and Research Questions

Approximately 3 years ago, an element of the Project and the focus of this study, the treatment terrace, received its first drop of runoff as a constructed wetland. This report describes this project's objectives and the primary questions that meeting those objectives will answer.

The goal of this project is to assess the pollutant removal performance of the CCWQP. The aim of this goal is to answer the research questions of this project and the objectives were chosen as short-term targets to reach this goal. From a design and construction standpoint, the CCWQP's objectives were divided into 3 categories, the third of which stated the goal of maximizing pollutant removal effectiveness. To do this, both gray and green infrastructure were designed and installed. First, a large (4.5 cfs/127.4 L/s treatment capacity; 30 cfs, 850 L/s maximum capacity) hydrodynamic separator, also known as a continuous deflective separator ("CDS unit"), was installed in the diversion pipe upstream of the terrace. For the green infrastructure element, a 2.2 ac (0.89 ha) constructed wetland was built to further treat effluent from the CDS unit through an in-series system of step pools, a retention pond, and 3 bioinfiltration cells (consisting of 79% pervious and 21% impervious soils)

My project, directly addressed the 3rd design goal, or “maximizing pollutant removal effectiveness.” The design contractors listed BMP effluent concentrations in their design documents as a way to benchmark the type of pollutant concentrations the constructed wetland was designed to achieve. However, the unique nature of the CW ultimately built complicates these comparisons, as the median effluent concentrations are typically for unique, standalone BMPs, not BMP elements built in series with pretreatment as the CCWQP’s CW is designed. Nonetheless, BMP effluent figures, coupled with Oregon Water Quality Criteria for many pollutants measured in this project can provide different lines of evidence for evaluating the pollutant levels received and leaving the CW.

Stormwater pollutant transport, transformation, and fate do not operate in a vacuum. Several interrelated biological, physical, and biochemical mechanisms drive these processes in these systems. The CW system under study was specifically designed to treat urban stormwater runoff, and the system is certainly loaded with considerable runoff from its industrialized catchment. Therefore, to get at understanding the CW’s effectiveness in reducing pollutant levels particularly across the terrace, concentration and mass-based reductions will be calculated for the 14 events to better understand how the system is transforming (e.g., sequestration, export, etc.) different pollutants. My main research questions presented in the Introduction guided my study objectives:

1. Evaluate pollutant concentrations of surface water leaving the treatment terrace at the CCWQP against ISWBMPdb benchmarks and Oregon Water Quality Criteria.
2. Assess spatial (e.g., before/after) and temporal (e.g., seasonal) trends in concentration-based and mass-based reductions on the treatment terrace.
3. Explore how precipitation could help qualitatively explain reduction differences.

Methods

Study design overview

A systematic sampling schedule was followed during the regulatory "wet" season (October-April) to capture pollutant concentrations and water flows during the period of the year when precipitation was most likely. As precipitation was hypothesized to be a driver of water quality and quantity in the CCWQP system, the wet season was targeted for sampling. Five separate monitoring points ("MP", Figure 11) across the project site were chosen to represent strategic points in the system.

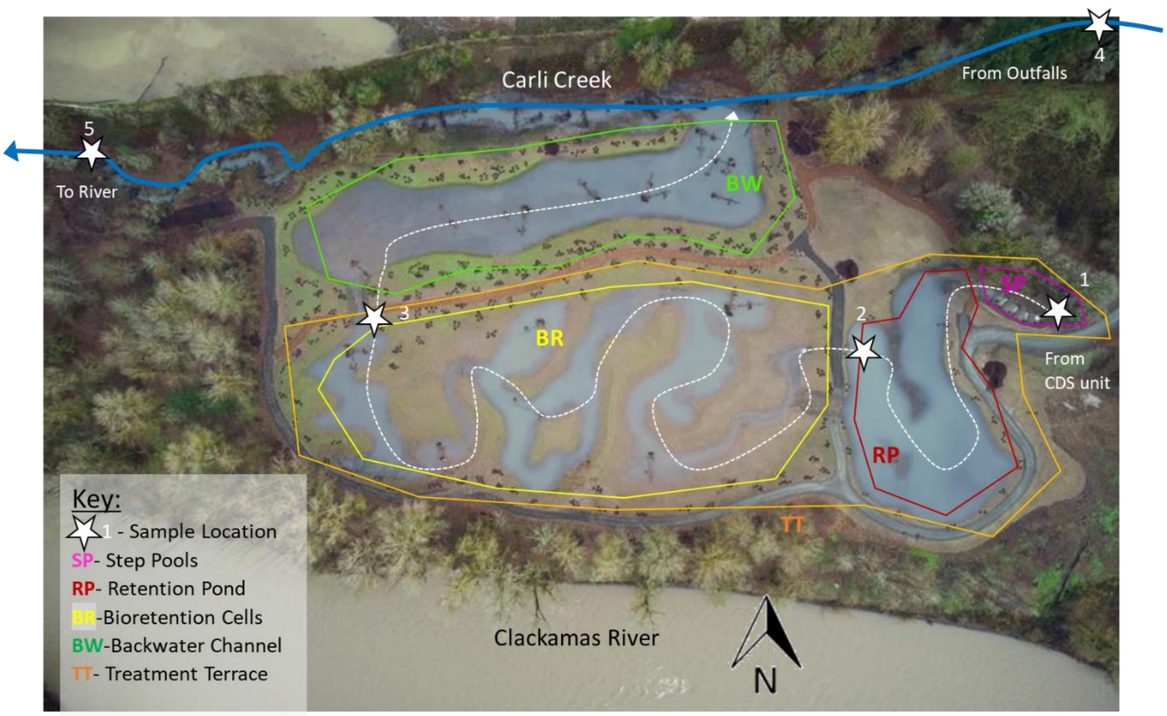

Figure 11. Monitoring Point map identifying sample locations at the CW.

Mointoring Point 1.	The 24" outfall from the hydrodynamic separator unit which falls into the step pools, was chosen to represent *influent* water quality and quantity entering the CW area of the project. Runoff at this site receives preliminary treatment of solids and floatables through the upstream treatment unit.
Mointoring Point 2.	The flow-control structure which modulates water from the Retention Pond to the bioretention cells was chosen to represent water quality at the effluent of this first BMP (*Post Retention*) in the series of the CW.
Mointoring Point 3.	The overflow berm at the outlet of the bioretention cells was chosen to represent final treatment terrace water quality and quantity (*Post Terrace*). Up until this point, it is assumed surface flows are hydrologically disconnected from the other features of the project. To

answer questions about pollutant reduction, pollutant loading at this site is compared with Monitoring Point 1.

Mointoring Point 4. This *upstream* site was chosen to represent typical, untreated MS4 outfall runoff from the highly industrialized catchment. No treatment occurs in this area and under low flow conditions (i.e., not backwatered), it is assumed this site is hydrologically disconnected from the downstream features due to Carli Creek's gradient and the large culvert.

Mointoring Point 5. This *downstream* site was chosen to represent final water quality conditions after flows from the treatment terrace flow through the backwater channel and are mixed with flows from Monitoring Point 4 on Carli Creek

Fourteen (14) separate events were scheduled to collect all water quality parameters. These were randomly scheduled within a week (avoiding weekends) but targeted to occur every two weeks. This number of events was chosen to balance resources (time, money) with the objective to obtain a representative picture of the water quality and quantity occurring at the site.

Environmental variables examined in this study were chosen as suspected drivers of runoff pollutant transport and transformation in the wetland. Precipitation in the catchment determined the loading of pollutants delivered to Carli Creek and the CW. While the treatment terrace is perched approximately 10 feet above the mouth of Carli Creek, a tributary to the Clackamas River, flow in the Clackamas River was also investigated under the hypothesis that a hydrologic connection existed between the treatment terrace and the Clackamas River, particularly during high wet season flows.

Pollutant selection

Stormwater pollutants were chosen based on several factors:

- Pollutants monitored in the course of CWES' fulfillment of its MS4 obligations
- An analysis and review of historic data at a long-term sampling site in the MS4 system, near 120^{th} and SE Carpenter St.
- A literature review of commonly found stormwater pollutants in urban runoff.

- Available benchmarks (International BMP database) and criteria (Oregon Water Quality Criteria) to compare pollutants against.
- Associated pollutants necessary to calculate certain Oregon Water Quality Criteria.
- Lab capacity/capabilities and cost per analysis.

A list of solids, nutrients, metals, and bacteria were chosen to represent pollutants of interest to CWES and those reasonably expected to be present in the MS4, in Carli Creek, and on the terrace. Table 6 below describes these pollutants and parameters.

Table 6. Pollutants and parameters of interest measured during this study.

General Chemistry	Nutrients	Metals	Field
Solids†	Ammonia	Cadmium*	Temperature
Hardness	Nitrate-nitrite	Copper*	pH
E. Coli	Total Phosphorous	Lead*	Dissolved Oxygen
		Zinc*	Conductivity
		Mercury, Total	

† = Total, Total Suspended, and Total Dissolved
* = Total and Dissolved

In general, analytical methods approved for Clean Water Act compliance (40 CFR 136) were used for all pollutants. The regulatory construct Clackamas County's MS4 system operates under is the Clean Water Act and monitoring conditions in CWES' current MS4 permit mandate use of 40 CFR 136, with rare exceptions. Analytical methods used for the parameters are listed in Table 7 below, including other pertinent details.

Table 7. Lab Parameters measured during the study.

Analyte	Units	Sample Matrix	Detection or Reporting Limit	Sample Prep Method	Analytical (Instrumental) Method	Performed by
Solids, Total Suspended	mg/L	Water	5	None	SM 2540-D	CWES Lab
Solids, Total Dissolved	mg/L	Water	5.6	None	SM 2540-C	CWES Lab
Solids, Total	mg/L	Water	5	None	SM 2540-B	CWES Lab

Analyte	Units	Sample Matrix	Detection or Reporting Limit	Sample Prep Method	Analytical (Instrumental) Method	Performed by
Hardness	mg $CaCO_3$/L	Water	5	None	SM 2340-C	CWES Lab
Total Phosphorous	mg P/L	Water	0.04	None	SM 4500-P A,B & F	CWES Lab
Ammonia	mg N/L	Water	0.02	None	SM 4500-NH3G	CWES Lab
Nitrate+nitrite	mg N/L	Water	0.03	None	SM 4500-NO3F	CWES Lab
Metals†	µg/L	Water	varied	Lab-Filter	EPA 200.8	CWES Lab
Total Mercury	µg/L	Water	0.2	None	EPA 245.2	Contract Lab

CWES Water Quality Lab (WQL), accredited by TNI/NELAC, provided complete analytical support for analysis of all lab parameters (except for field parameters such as pH, DO, conductivity, and temperature). Field parameters were measured by the methods listed below in Table 8.

Table 8. Field Parameters measured during the study.

Analyte	Units	Sample Matrix	Resolution	Analytical (Instrumental) Method
Temperature	°C	Water	0.1	SM 2550-B
pH	S.U.	Water	0.01	SM 4500-H B
Dissolved Oxygen, DO	mg/L, % sat.	Water	0.1	SM 4500-O C EPA 360.1
Conductivity	µS/cm	Water	0.1	SM 2510-B

The author, upon logging samples into the WQL's Laboratory Information Management System (LIMS), relinquished sample bottles for laboratory analysis. As necessary, WQL staff arranged for contract lab courier pick-up for certain analyses and analyzed the remainder of lab samples in-house. WQL staff would conduct a Quality Assurance (QA) review of analytical data generated following an internal Quality Assurance Manual (QAM). Upon validation, the lab would issue hard copy reports with results. The LIMS system could also be directly queried.

Benchmark and Criteria Selections

In this report, benchmark refers to effluent median concentrations of different stormwater BMP design elements as published in the ISWBMPdb (2020). This database is a long-term research effort led by the Water Research Foundation. The purpose of the database is to provide data to practitioners, scientists, and policymakers to improve the use and functional understanding of stormwater BMPs in the real world. For the purposes of this project, pollutant concentration data collected at Site 2 (*Post Retention*) were compared to median effluent concentrations for the BMPs "Retention Pond" and "Wetland Basin." The ISWBMPdb defines a Retention Pond as a "surface wet pond with a permanent pool of water…" and a Wetland Basin as a "similar to a retention pond (with a permanent pool of water), typically with more than 50 of its surface covered by emergent wetland vegetation." The retention pond at the CCWQP resembles both of these BMP definitions therefore comparisons were made with both.

Pollutant concentration data collected at Site 3 (*Post Terrace*) were compared to median effluent concentrations for the BMP "Bioretention." The ISWBMPdb defines bioretention as "Shallow, vegetated basins with a variety of planting/filtration media and often including underdrains. Also called rain gardens and biofiltration." This definition very closely matches the design and construction of the bioretention cells on the treatment terrace.

Current Oregon Water Quality Standards were used to evaluate other specific pollutants. In all cases, only Monitoring Points 4 and 5 were assessed as they were located within Carli Creek. For Total Dissolved Solids, the Willamette River guidance value was chosen as the Clackamas River is a tributary to the Willamette and Carli Creek is a tributary to the Clackamas (OAR 340-041-0345 (2)). For *E. coli*, the freshwater contact designated use was chosen to evaluate water quality against the criteria (OAR 340-041-0009). For metals (i.e., cadmium, copper, lead, and zinc), the current Toxics Standards (OAR 340-041-0033) were used to assess attainment. In the case of cadmium, lead, and zinc, hardness-dependent criteria are used to calculate acute and chronic exposure criteria. The chronic criterion for each metal was always more stringent, so only that criteria was calculated for this study, even though criteria were calculated for each event in order to investigate events individually. Oregon has adopted the Copper Biotic Ligand Model (“BLM”, Oregon Department of Environmental Quality [ODEQ], 2022) for its freshwater aquatic life copper water quality criteria. This toxics standard is an instantaneous criteria that requires collecting 11 different water quality parameters to derive point-in-time acute and chronic criteria for dissolved copper. All 11 parameters were not collected in the course of this study at each Monitoring Point. Of the parameters that were collected (i.e., temperature, pH, hardness, conductivity), they were used directly in the BLM calculation software or used to derive other values. Where other parameters were not available, region-specific default values were chosen to calculate the acute and chronic criteria. Due to this, these criteria are conservative estimates of Oregon water quality criteria for dissolved copper.

Flow monitoring set-up

In order to calculate pollutant loading, water quality data with corresponding water quantity data is required. However, due to the different flow regimes at each of the MPs, different approaches were made to attempt to estimate flow throughout the sampling campaign.

Monitoring Point 1 is a piped stormwater outfall. A battery-powered Hach FL900AV flow meter/datalogger and submersible area-velocity sensor was installed to measure flow. This type of equipment requires specific flow behavior (i.e., direction changes, hydraulic jumps, etc. alter velocity profile, producing inaccurate flow data) and typically has a site-specific ceiling for velocity measurements (i.e., it can only accurately measure up to about 9 feet/second (2.74 meters/second). These equipment limitations meant the equipment was installed 2 manholes up

(See Figure 12) but below the hydrodynamic separator. The equipment was installed and calibrated in the field initially. Periodic checks to download data occurred throughout the project.

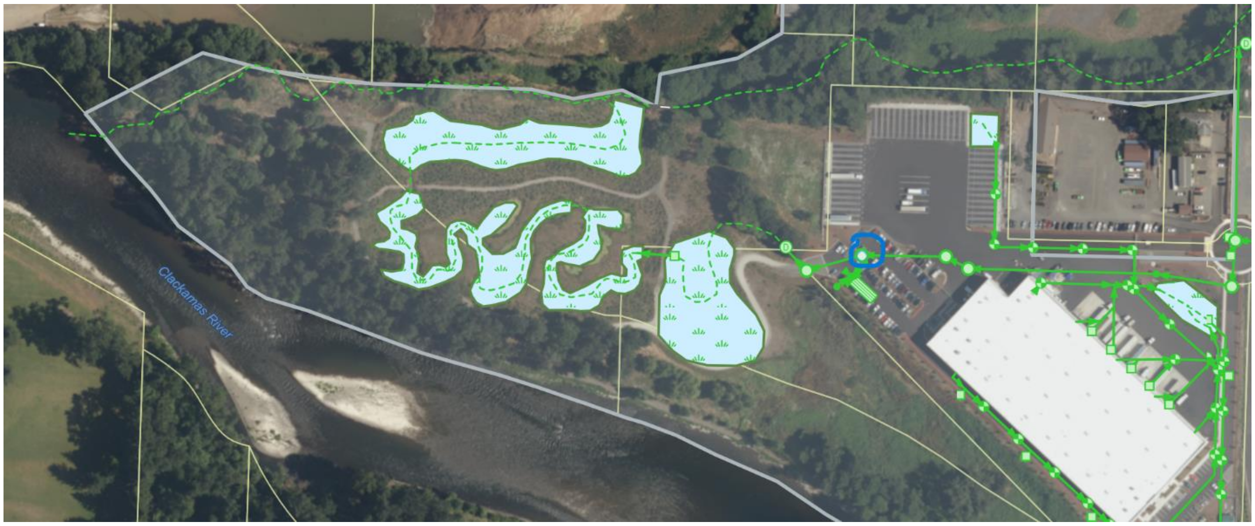

Figure 12. Installation of Monitoring Point 1 flow meter and sensor relative to the CCWQP.

Water quantity was also desired at the other 4 MPs to calculate detailed pollutant reductions across the project MPs but was deemed unnecessary or impractical at Points 2 and 5. Point 2 had a piped section (8 in/ 20.3 cm) which could theoretically have had a level or area-velocity sensor installed. However, there existed a bermed overflow which was designed to bypass the flow-control structure when the water level in the pond reaches a specific elevation. Bypassing flow would be unaccounted for in high flow scenarios so installation of flow monitoring equipment at Point 2 was abandoned. Point 5 presented a possible stage-discharge location but was deemed to be a poor location due to frequent backwatering occurring during high Clackamas River flows (the location back-watered twice during the sampling campaign). Back-watered locations do not produce accurate discharge records using stage-discharge stations because of the site conditions necessary to produce reliable, long-term stage-discharge relationships. While advanced equipment capable of overcoming this obstacle and measuring discharge at this site was not installed due to resource constraints, a staff gage was installed and depth measurements were made during sampling events. When flows were low (i.e., water depth fell onto a measurable stage reading), occasional discharge measurements were made.

At both MPs 3 and 4, stage-discharge stations were constructed. For Point 3, a solar-powered datalogger and vented pressure-transducer system was installed to accompany a staff gage. Point 4, due to the density of tree coverage, small battery-powered self-recording vented pressure-transducer accompanied by a staff gage. Equipment procurement and infrastructure installations were completed under the guidance and support of Jeff Budnick, staff Hydrologist for WEST Consultants Inc., a CWES consultant. Periodic discharge measurements were conducted following standard USGS methods. A 6 ft top-setting wading rod in conjunction with a calibrated Marsh-McBirney electromagnetic velocity sensor were used to conduct cross-section discharge measurements. A stage-discharge curve was created for each site, with the point-of-zero-discharge (a stage reading signifying no discharge) determined at installation.

Water Quality Monitoring

Water Quality monitoring conducted to generate concentration results for comparison with benchmark and water quality criteria as well as for loading calculations. With the exception of field parameters (i.e., pH, DO, temperature, and conductivity) and *E. coli*, all parameters were collected as 24-hour time-proportional composites using Hach AS950 portable autosamplers deployed in the field (Figure 13). This, along with flow measurement strategies, are summarized below (Table 9).

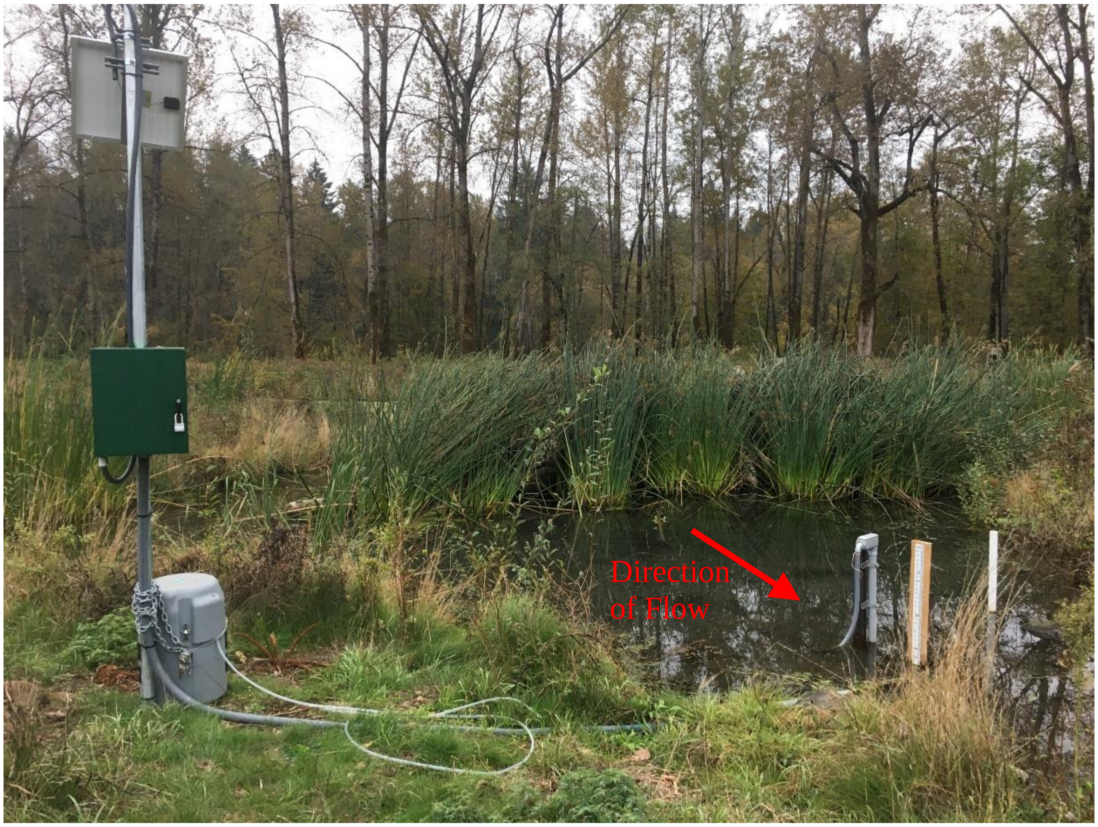

Figure 13. Portable autosampler being set-up at Monitoring Point 3.

Table 9. Sample types for water quality parameters.

Parameters	Monitoring Points 1	2	3	4	5
Grab					
Temp, DO, Conductivity, pH, *E. Coli*	✔	✔	✔	✔	✔
24-hour composite					
Solids, Hardness, Nutrients§, Metals†	✔	✔	✔	✔	✔
Total Mercury	✔	✔	✔	✔	✔
Flow					
Measurement strategy*	FS	E	SD	SD	E
Continuous or **D**uring sampling events only	C	D	C	C	D

§ = Total Phosphorous, Ammonia Nitrogen, and Nitrate+Nitrite Nitrogen
† = Total and Dissolved Cd, Cu, Pb, and Zn
* = Flow sensor (FS), One-time Discharge estimate (E), Flow meter/Sensor (FS)

E. coli grab samples were collected by dipping directly into the surface water. Field parameters were measured by collecting a ~500 mL sample in a clean beaker and measuring using YSI Professional Plus Multimeter connected to a Quattro cable equipped with pH, conductivity/temperature, and Dissolved Oxygen sensors. The sensors were calibrated prior to each event following 40 CFR 136 methods.

Composite samplers were programmed to collect sub-samples every 20 minutes and stop sampling after 24-hours. Intake tubing and strainers were routed to and secured about 50% up the water column on installed staff gages (for Points 3, 4, and 5) or left to sit on the bottom of the piping (Points 1 and 2) to avoid sampling excessive sediment. Due to technical errors (e.g., defective battery, mis-programming), occasionally a sampler would fail to collect the full 24-hour composite. In these instances, a single grab was collected on the second day and sub-sampled into the appropriate bottles for analytical parameters.

Environmental Predictors

Environmental variables expected to explain variability in pollutant reduction ("predictors") included precipitation and Clackamas River flow. The nearest precipitation gage maintained by CWES and the USGS-operated flow gage are both shown in the figure (Figure 14) below. Precipitation patterns are very complex in northwest Clackamas County due to a number of factors (e.g., topography, wind, etc.) so the closest precipitation gage was chosen to represent rainfall at the site. Eda Creek is the only tributary between Carli Creek's confluence with Clackamas River and the USGS gage. Via telemetry, 15-minute data is transmitted to online portals every hour or so. This data was downloaded periodically.

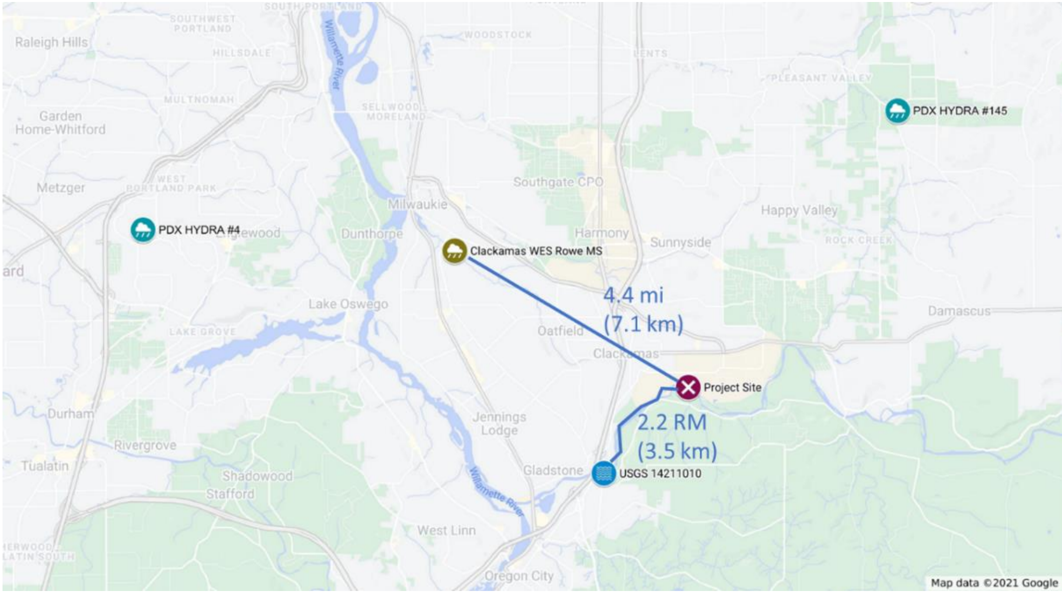

Figure 14. Geographic location of environmental variables for this project.

Data Processing and Reduction

In order to evaluate the water quality and environmental data gathered during this project, several processing steps were conducted prior to generating statistics and visualizing data. A data map, describing these processing steps, is shown in the figure below (Figure 15). To handle water quality data with "less than" values, an absolute value of ½ of the reporting limit was used (e.g., a TSS result reported as <5.0 mg/L was changed to 2.5 mg/L). Specifically for *E. coli*, where results could be "greater than" a value, due to inadequate volume sampled to conduct dilutions, or "less than" the detection limit, the reported values were used. In other words, for a result of >2420 MPN/100 mL, a value of 2420 MPN/100 mL was used in calculations. Also unique to *E. coli* was the calculation of a geometric mean to represent the central tendency of measurements versus an arithmetic mean. This is common practice in reporting indicator bacteria sample sets in regulatory settings and is calculated using Equation 1.

$$\left(\prod_{i=1}^{n} a_i\right)^{\frac{1}{n}} \qquad \text{(Equation 1)}$$

Where Π = denotes a series of multiplications for each sample in the set
n = number of samples
i = the *i*th sample

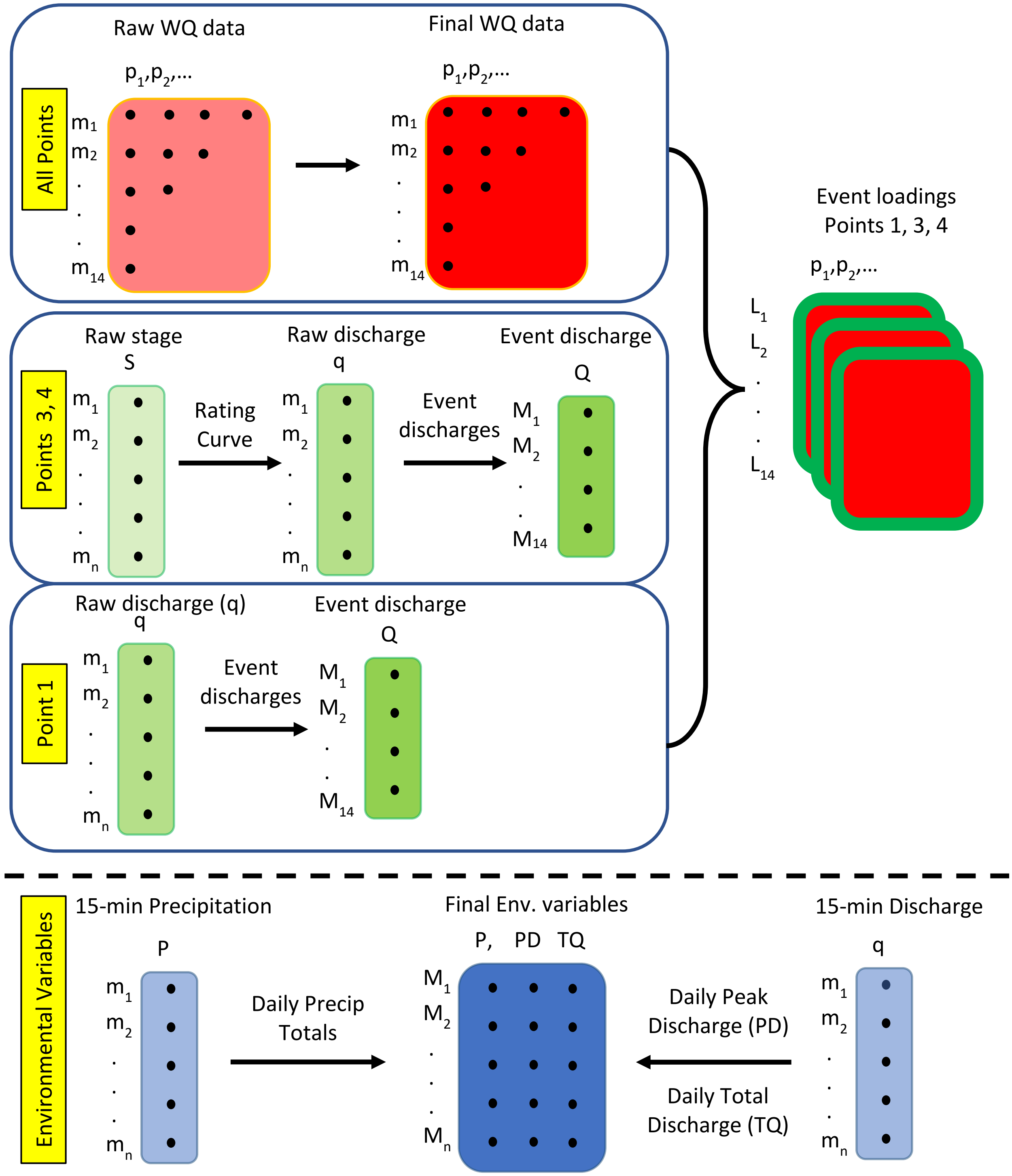

Figure 15. Data map describing how raw field data was processed.

Concentration and Loading reductions were calculated using the equations (Equations 2, 3, and 4) shown below. These were chosen to not only compare and contrast the different approaches to interpreting "reduction" (i.e., on a concentration- versus loading-basis), but also as they are commonly used and typically deemed adequate to understand pollutant fate and transport in stormwater BMPs (Lenhart & Hunt, 2011; Carleton et al., 2000). Median reductions are commonly published as representing overall performance, however individual reductions were evaluated here to study the temporal change over the sampling campaign. In these calculations, Monitoring Point 1 (*Influent*) is considered the “inflow” to the treatment terrace as this is the primary input to the system. Monitoring Point 3 (*Post Terrace*) is considered the “outflow” of the treatment terrace, as the point captures the dominant surface flow volume from the treatment terrace. Therefore, in this analysis, the system is considered the treatment terrace and the in-series elements of a retention pond and bioretention cells. The backwater channel is separated as it was designed to (and frequently does) capture high flows from storm runoff entering Carli Creek from the MS4 network.

$$\boldsymbol{Percentage\ Concentration\ Reduction} = \frac{[Inflow_x] - [Outflow_x]}{[Inflow_x]} * 100\% \qquad \text{Equation 2}$$

$$\boldsymbol{Percentage\ Mass\ Reduction} = \frac{Inflow_x,\ (mass) - Outflow_x,\ (mass)}{Inflow_x,\ (mass)} * 100\% \qquad \text{Equation 3}$$

$$Mass_x(lbs) = [Inflow_x] * constant * discharge \qquad \text{Equation 4}$$

Where: $[\mathrm{Inflow}_x]$ = The influent concentration of pollutant x
$[\mathrm{Outflow}_x]$ = The out-flowing concentration of pollutant X
mass units = pounds
Constant = 8.34
Discharge = Million gallons/day

As written, positive concentration and mass reductions correspond to decreases in pollutant levels from Monitoring Point 1 to Monitoring Point 3. Negative reductions correspond to the opposite (i.e., an increase, or pollutant export). Concentrations were entered from analytical data collected from the 24-hour time-proportional composite samples which represent the average, time-weighted concentration of each pollutant over a single day. Discharge volumes were calculated over the same period of time as the 24-hour composite period representing concentrations.

Specifically for *E. coli*, rather than concentration-based reductions, log-reductions were calculated as they are frequently used for expressing reductions in bacteria due to some external treatment (e.g., in wastewater plants, treatment is typically disinfection via UV exposure, gaseous chlorine, etc.). The formula for calculating log-reductions is given in Equation 5 below. A log reduction of 1 would be a 90% reduction in the number of organisms, 2 would be a 99% reduction, etc.

$$\boldsymbol{log\ Reduction} = \ log_{10}\left(\frac{N_0}{N}\right) \qquad \text{Equation 5}$$

Where: N_0 = Initial number of organisms
N = Final number of organisms

Sampling Plan

Prior to initiation of the any equipment installation or field work, a Quality Assurance Project Plan ("QAPP") was drafted (Lombard & Kirchmer, 2004). This Plan served two purposes: 1) to clarify project goals and tasks and 2) effectively communicate the project's schedule, budget, and outcomes to stakeholders. In the scope of this work, the QAPP doubled as a Sampling and Analysis Plan ("SAP"), which monitoring programs often separate. Within the document, the specific quality assurance objectives are detailed. Of note are the 3 project-specific quality control samples collected in addition to the events: 2 field duplicates and one equipment blank. Target precision metrics were adopted from state volunteer monitoring guidance (ODEQ, 2021). The QAPP is not detailed here but can be found in Appendix A. Of note is that many elements were changed through the course of the project due to feasibility, objective refinement, and time and resource constraints.

Results

Overview

Fourteen events were successfully sampled over the sampling campaign. All field and analytical parameters were collected and analyzed for each event, at each monitoring point.

For nearly all of the events, the portable autosamplers successfully completed their 24-hour time-proportional composites as programmed (Table 10). This resulted in a "success rate" (successful composite sampling events multiplied by MPs) of 96%.

Table 10. Summary table of all monitoring events and notable comments.

Event	Start Date	Comments
1	10/12/2020	Monitoring Point 5 sampler failed. All grabs successful.
2	10/28/2020	Monitoring Point 3 sampler failed. All grabs successful.
3	11/9/2020	Monitoring Point 4 sampler failed. All grabs successful.
4	11/23/2020	
5	12/9/2020	Vegetation cut back & removed around Retention pond prior to event
6	12/21/2020	Retention pond overflowing the design berm. Lower Carli creek, Monitoring Point 5 backwatered (i.e., high Clackamas River flows).
7	1/6/2021	Retention pond overflowing.
8	1/20/2021	Field Duplicate collected at Monitoring Point 3
9	2/8/2021	
10	2/17/2021	Retention pond overflowing the design berm.
11	3/3/2021	
12	3/15/2021	
13	3/31/2021	Field Duplicate collected at Monitoring Point 3
14	4/12/2021	Equipment Blank collected

While field and analytical parameter sampling was largely successful, flow equipment set-up and measurement was delayed and not begun until the 4th event. Several equipment malfunctions also left gaps in the flow record, particularly at Monitoring Point 1, which did not allow calculation of flow reduction and pollutant reductions by mass for certain events. On events 9 and 10, the velocity sensor on the submersible area-velocity sensor for Monitoring Point 1 failed, only providing water depth data. This water depth data was used to calculate flow via Manning's Equation with a roughness coefficient of n=0.012 (American Concrete Pipe Association, 2011), a slope of 0.0143, and a pipe diameter of 1.5 ft (0.46 m). Nonetheless, pollutant masses were calculated for 10 out of 14 events, or 71%.

Campaign quality assurance (QA) and quality control (QC) samples also showed good performance in terms of reproducibility and equipment cleanliness (Appendix B). Parameters for both field duplicates were within target RPD values, except for total and dissolved zinc and ammonia-nitrogen on Field Duplicate 1. No apparent issues with sampling were observed which could explain this high variability. The campaign equipment blank was clean (i.e., all laboratory parameters were less-than their detection limits), except for dissolved copper and zinc and total zinc. The dissolved copper results were just above the analytical detection limit of 0.10 µg/L, while the total and dissolved zinc results were approximately 2X the detection limit. Zinc is a very challenging metal to clean and exclude in sampling, however, these elevated background zinc levels suggest measured zinc in field samples are elevated beyond true environmental levels.

An aggregate summary of historical data (2008-2020, n=114) at a location upstream of the 54” outfall at the intersection of SE Carpenter Dr and 120th St is shown in Table 11 below. Until 2018, some sampling events specifically targeted storm (rainfall >0.10 inches, 25 mm) conditions, reflecting higher concentrations of pollutants in general than other events collected during dry weather conditions. The data in Table 11 was pre-processed by computing less-than values to ½ the detection limit and retaining greater-than values as the value. Distributions were left-skewed with many outliers for all but most of the field variables (Table 11) which were normally distributed. The high variance reflects the storm-targeted monitoring approach taken at CWES through the years for MS4 monitoring at the historic Carli Creek site.

Table 11. Aggregate data summary of historical site at SE 120th Ave and Carpenter Dr.

Variable	Units	Min	Max	Median	$\bar{x}$	σ
pH	S.U.	5.5	7.8	6.9	6.9	0.44
Temperature	° C	3.9	21.3	12.7	13.1	3.10
Dissolved Oxygen	mg/L	4	12.2	9.4	9.1	1.50
Conductivity	µS	5.98	534	193.15	177.9	99.67
E. coli	MPN/100 mL	1	2420	34.5	323.1	640.31
Hardness	mg $CaCO_3$/L	2.5	175	75.5	72.2	38.07
Total Solids	mg/L	13	341	144	141.1	60.96
Total Suspended Solids	mg/L	0.5	62	5	9.6	11.55
Total Dissolved Solids	mg/L	0.5	276	132	119.4	58.20
Ammonia Nitrogen	mg N/L	0.025	0.17	0.025	0.0	0.02
Nitrate-Nitrite Nitrogen	mg N/L	0.045	4.1	0.93	0.9	0.53
Ortho-phosphate	mg P/L	0.005	0.12	0.05	0.0	0.03
Total Phosphorous	mg P/L	0.02	1.27	0.05	0.1	0.12
Total Copper	µg/L	0.7	14.1	1.815	2.7	2.19
Dissolved Copper	µg/L	0.05	4	0.725	1.0	0.71
Total Lead	µg/L	0.034	19.8	0.355	1.1	2.16
Dissolved Lead	µg/L	0.005	1.79	0.04	0.1	0.18
Total Zinc	µg/L	11	129	25	34.4	24.03
Dissolved Zinc	µg/L	7	112	15.8	22.0	15.51

Environmental variables: Precipitation, Weather, and Clackamas River Flow

Precipitation during the start of the 2021 water year (October 2020-September 2021) was above average compared to the, albeit short, historical record (September 2017 – October 2020) of the Rowe Middle School rain gage (Figure 16). An unusually large rain event occurred over 4 days (January 10-14, 2021, total precipitation=3.42 in (86.9 mm)), which was preceded by very wet conditions spanning several weeks. The bulk of the precipitation occurred on January 13, 2021. A figure of the Clackamas River discharge is also shown in Figure 17.

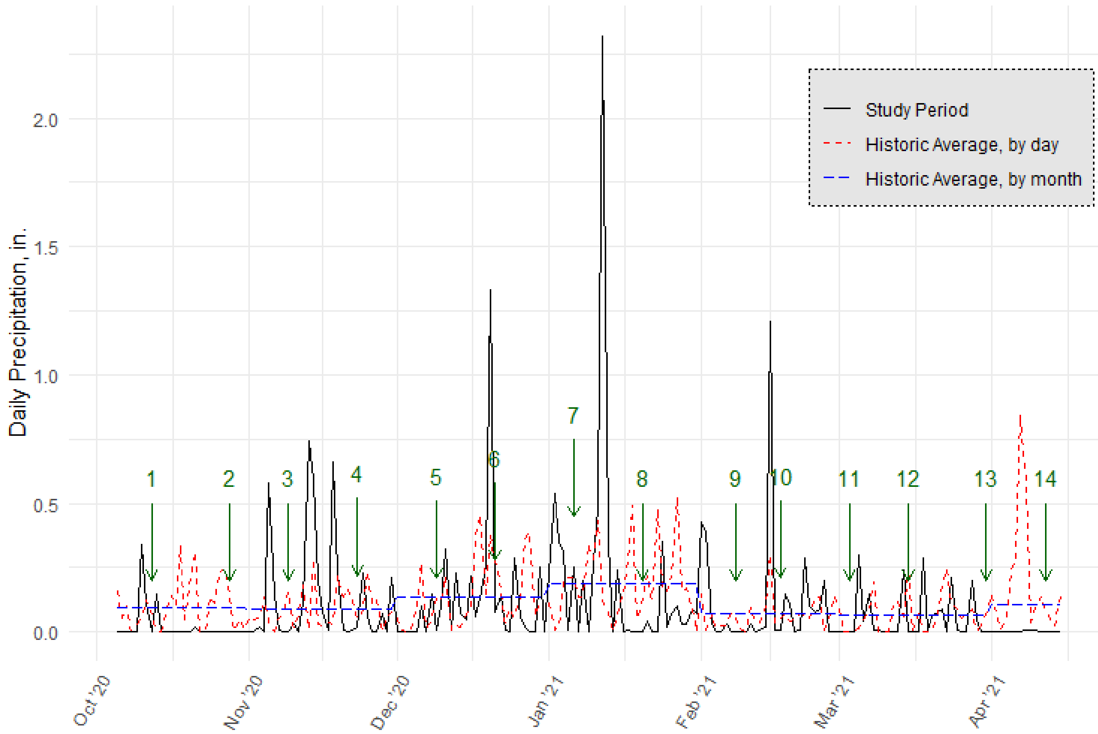

Figure 16. Precipitation during sampling campaign with historic average daily totals plotted by day (red dash) and historic average daily total by month (blue dash). Historic period from September 2017 to September 2020, study period excluded. Green arrows indicate sampling event.

Some summary statistics of the precipitation patterns around all events are given in Table 12 below. Also noteworthy was that the historic ice storm in February 2021, which left over 100,000 Clackamas County residents without power (Gormley, 2021), also occurred during the sampling campaign.

Table 12. Precipitation-based environmental values for each sampling event.

		Count of Antecedent		Intra-event Precipitation		Total Precipitation prior to event, in.	
Event	Start Date	Dry* Days	Wet* Days	Total, in.	Peak, in./hr	48 hours	72 hours
1	10/12/2020	0	2	0.14	0.07	0.76	1.55
2	10/28/2020	15	0	0.00	0.00	0.00	0.00
3	11/9/2020	1	0	0.05	0.01	0.12	0.28
4	11/23/2020	0	1	0.1	0.05	0.13	0.13
5	12/9/2020	0	1	0.02	0.01	0.4	0.42
6	12/21/2020	0	3	0.41	0.16	2.82	3.16
7	1/6/2021	1	0	0.46	0.13	0.62	1.09
8	1/20/2021	5	0	0.08	0.01	0	0
9	2/8/2021	5	0	0	0	0.06	0.06
10	2/17/2021	1	0	0.01	0.01	1.97	2.11
11	3/3/2021	4	0	0	0	0	0
12	3/15/2021	0	1	0	0	0.25	0.25

		Count of Antecedent		Intra-event Precipitation		Total Precipitation prior to event, in.	
Event	Start Date	Dry* Days	Wet* Days	Total, in.	Peak, in./hr	48 hours	72 hours
13	3/31/2021	2	0	0	0	0.05	0.26
14	4/12/2021	13	0	0	0	0	0.02

*Only whole days were counted. A "Dry" day had <0.10 inches of rain and a "Wet" day had ≥0.10 inches of rain.

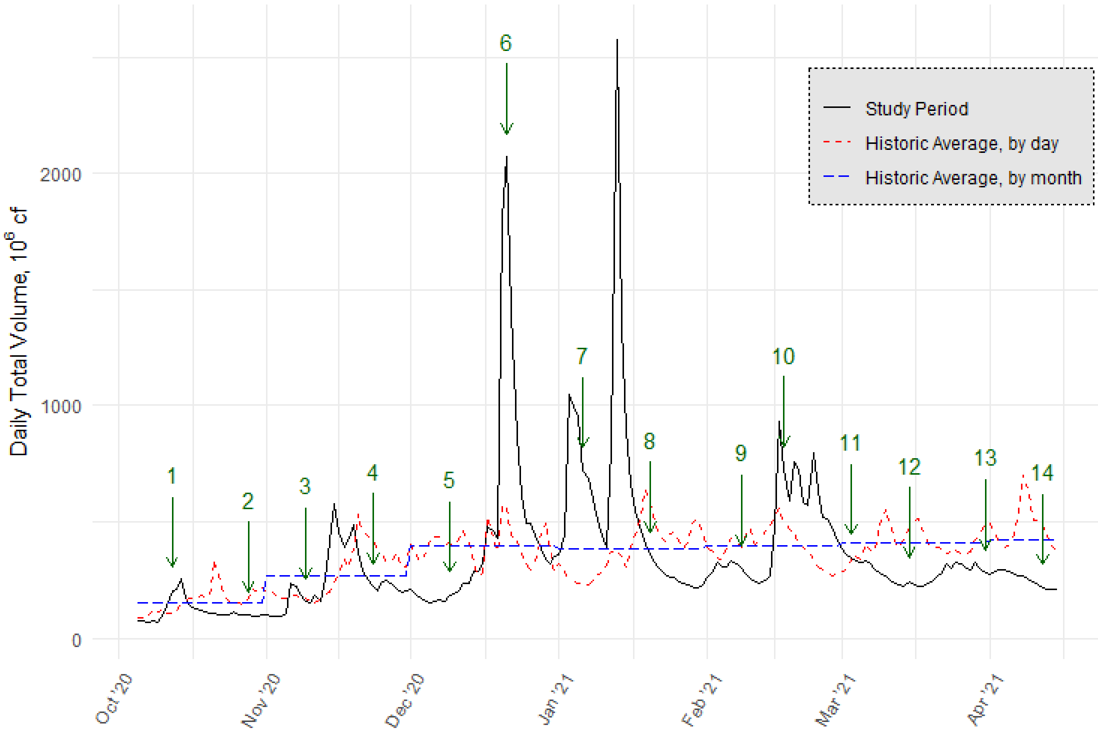

Figure 17. Clackamas River total daily flow at USGS station 14211010 during sampling campaign with historic average daily totals plotted by day (red dash) and historic average daily total by month (blue dash). Historic period from September 2010 to September 2020, study period excluded. Green arrows indicate sampling event.

Flow

Flow data at MPs 1 and 3 were measured directly and were used in conjunction with concentration data to calculate pollutant mass for individual events, where equipment failures did not affect data quality. A continuous flow record was also collected at MP 4 (*Carli Creek upstream*). Available flow data for the study period is not included here but a window from December 1, 2020 through February 1, 2021 is shown in Figure 18 below, which shows the effect of infiltration on the treatment terrace during these winter storm flows. Also evident are the generally lower "baseline" discharge rate at MP 3, and the longer "tail"/return-to-baseline compared to MP 1 after each storm event, symbolic of a slow release of the storm flows after each peak.

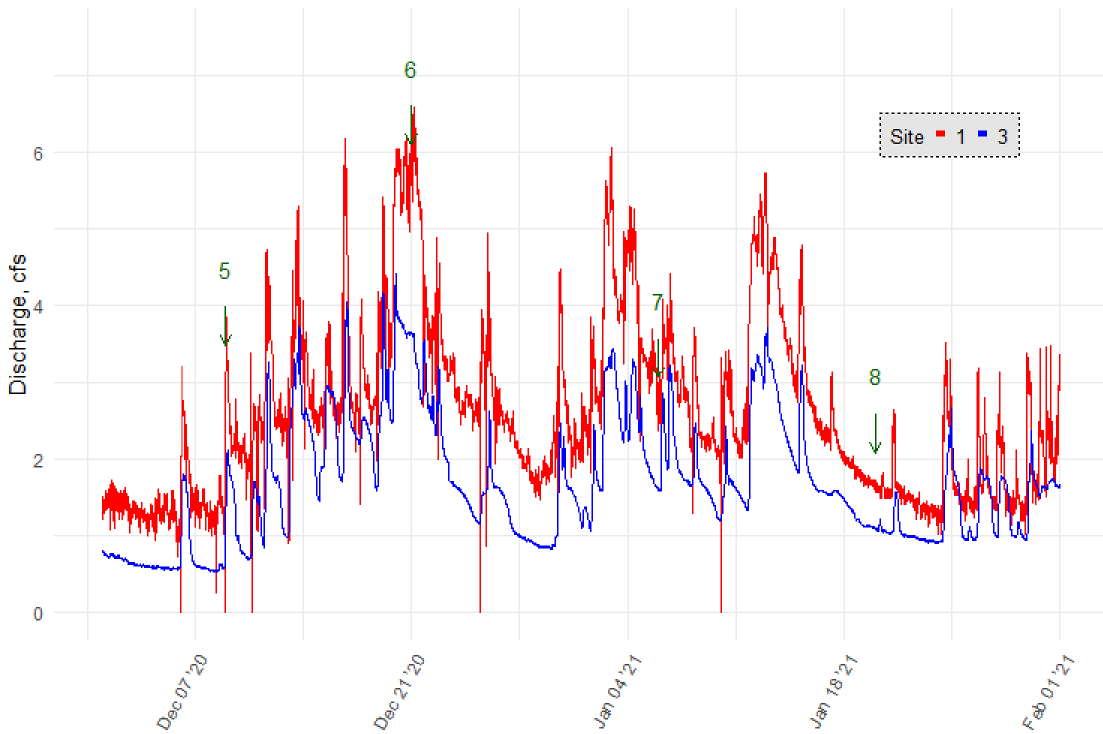

Figure 18. Flow rates at Monitoring Point 1 and 3 in the Project. Sampling events identified with green arrows.

An analysis of the resolvable (n=30) storm peak flows throughout this limited window of the sampling campaign (December 1, 2020 – February 1, 2021) was performed to estimate peak flow reductions by the treatment terrace (i.e., peak flow reduction between Monitoring Point 1 and 3), although this metric wasn't a focus of this study. Considering peak discharge measurements were not made in developing the rating curve (Appendix C) for site 3, actual peak flow reductions are likely higher. This is because the rating curve likely underestimates discharge at high stages, causing peak flow reduction estimates on the treatment terrace to be conservative. That said, peak flow reductions averaged 35.7% (σ = 7.26%) during this window.

An analysis of flow volume reductions during 10 of the events (conducted from November 23, 2020 through March 31, 2021) showed an average volume reduction of 345,000 ±

gal. (1,314,000 L), or 28%. Associated peak flow reductions was 19%. Two (on February 17, 2021 and March 13, 2021) of these events had net increases of flow volume (Figure 19).

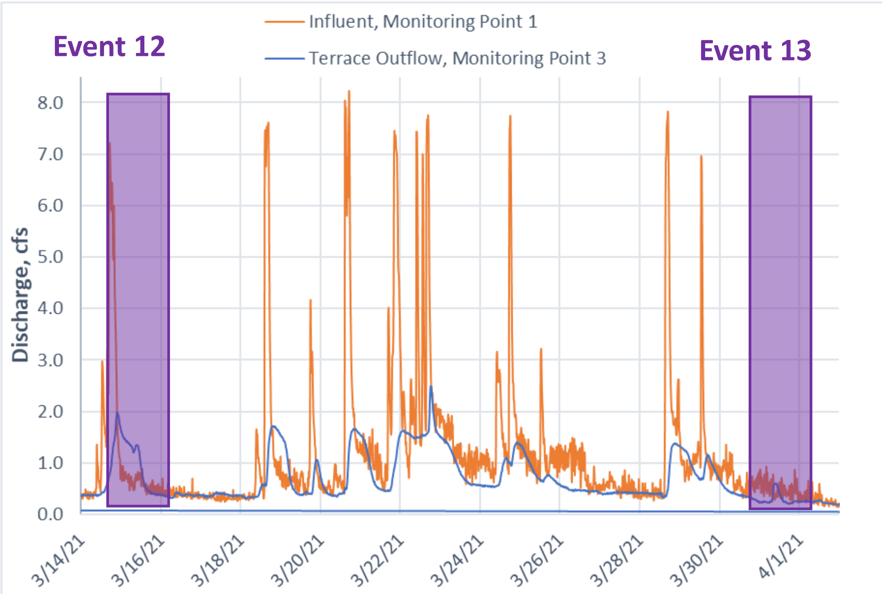

Figure 19. Discharge at MPs 1 and 3. Hydrograph shows distinct lag between discharge peaks, and extended "tail" of peak.

The Project studied here was very effective at minimizing peak flows and in many cases, delaying the onset of peak flows (Figure 19), as seen by the discharge curves comparing MP 1 and 3. In the context of this study, there were two events where the calculated volume at MP 3 was higher than at MP 1 (see Event 12 in Figure 10 above), suggesting more water leaving MP 3 than entering at MP 1. Removing these events, average volume reductions were 491,000 ± 186,000 gal (1,867,000 ± 699,000 L) and average peak flow reductions were 38 ± 11 %. These peak flow reductions were similar to results from the limited time-frame analysis conducted above. The net increase in flow is likely due to the selection of the time window for measuring flows, which was chosen to coincide with the 24-hour composite sampling window. There is a noticeable "lag" in inflow to the terrace and outflow into the backwater channel, and the two events with net increases occurred such that the retained volume which was released in the time window at MP 3 was greater than what flowed into at MP 1.

This is an example of the terrace's hydraulic capacity on the terrace. This "lag" feature was an unstudied element and study design limitation for this Project. This delay was not accounted for in concentration- and mass-based calculations for separate pollutants, the effect on both likely being a function of flow at MP 1, antecedent dry conditions, existing storage capacity of the terrace, and other environmental variables. Future calculations should incorporate CW hydraulic residence times when determining volume at different points in the constructed wetland.

Field Parameters

Field parameters were measured throughout the study period at all five sites. A statistical summary is given in Table 13 below. Average and median dissolved oxygen grab measurements were above 6.1 mg/L at all sites but tended to be highest at Monitoring Point 1. Dissolved oxygen at the two instream sites (MPs 4 and 5) were above the Oregon Cold-water temperature dissolved oxygen criteria of 8.0 mg/L for a 30-day mean (OAR 340-041-0016). Temperature typically decreased from the inflow to the outflow of the terrace and from the upstream Carli Creek site to the downstream site.

Table 13. Summary Statistics for field parameters during the studying period at the 5 MPs.

Parameter, Units	Statistic	Monitoring Points				
		1	2	3	4	5
Temperature, ° C	$\bar{x}$, Median	12.6, 11.9	11.2, 10.4	9.19, 9.25	10.1, 9.95	8.26, 8.5
	σ, Range	1.8, 10.4–15.9	1.9, 9.0–15.6	2.2, 5.8–15.0	1.9, 7.5–15.5	2.4, 4.6–4.8
Dissolved Oxygen, mg/L	$\bar{x}$, Median	10.2, 10.2	9.18, 9.25	9.5, 9.85	9.78, 9.7	9.79, 10
	σ, Range	1.3, 8.6–12.5	0.96, 7.4–10.7	1.4, 6.1–11	0.90, 8.1–11.4	0.90, 8–11.1
Conductivity, µS/cm	$\bar{x}$, Median	149, 157	137, 141	131, 133	133, 136	144, 142
	σ, Range	39, 82.2–238	39, 77.9–227	32, 69.2–193	21, 97.7–176	32, 98.3–203
pH, S.U.	$\bar{x}$, Median	7.67, 7.71	7.41, 7.46	7.34, 7.31	7.24, 7.24	7.28, 7.25
	σ, Range	0.23, 7.17–8.01	0.28, 6.81–7.93	0.44, 6.48–8.34	0.32, 6.43–7.66	0.34, 6.47–7.92

In comparison to historic values at an outfall in the MS4 system upstream of Monitoring Point 1, temperature and conductivity means were nearly equivalent (historic $\bar{x}$: 13.1° C, 177.9 µS/cm). Historic mean pH dissolved oxygen values however were on average higher however (historic $\bar{x}$: 6.9 S.U., 9.1 mg/L).

Solids and Hardness

Solids and Hardness were measured throughout the study period at all five sites. A statistical summary is given in Table 14 below. During most events, Total Suspended Solids (TSS) results were less than the detection limit. Therefore, a summary of these "less than" values is also given. Other statistics (e.g., mean and median) are calculated using the ½-the-detection limit rule for TSS. Total Dissolved Solids (TDS) was typically the bulk of the solids fraction, during each event, as evidenced by the TDS and Total Solids (TS) statistics being very similar and the low TSS results. For the inflow to the terrace (MP 1), low TSS is reflective of a functioning CDS unit upstream which, when properly maintained, is effective at removing large solids delivered from the MS4 system.

Table 14. Summary Statistics for solids and hardness.

Parameter, Units	Statistic	Monitoring Points				
		1	2	3	4	5
Total Suspended Solids, mg/L	$\bar{x}$, Median	5.0, 2.5	4.8, 2.8	11.8, 2.5	3.5, 2.5	5.5, 5.5
	σ, Range	5.2 2.5–18.0	2.9, 2.5–9.0	28.7, 2.5-111	2.8 0.5–12.0	3.7, 0.5–14.0
	# non-detects, % of total	9, 64%	7, 50%	8, 57%	11, 79%	6, 43%
Total Dissolved Solids, mg/L	$\bar{x}$, Median	141, 137	129, 141	114, 119	124, 136	130, 140
	σ, Range	36.8, 64–206	34.6, 63–176	33.8, 64–173	23.6, 79–	30.4, 79–162
Total Solids, mg/L	$\bar{x}$, Median	143, 141	144, 143	135, 135	134, 132	140, 144
	σ, Range	34.8, 87–211	39.2, 74–214	55.3, 62–276	25.8, 95-176	31.2, 78–183
Hardness, mg $CaCO_3$/L	$\bar{x}$, Median	76.1, 79	75.7, 80.5	68.0, 70.5	76.5, 81.5	79.6, 77.0
	σ, Range	21.9, 31–114	23.8, 30–116	25.2, 33–111	19, 42–107	23, 43–112

With the exception of the final event, all TSS results at MP 3 were below the ISWBMPdb median effluent value for bioretention BMPs (Figure 20). Furthermore, *all* TDS results were below the median effluent value for bioretention BMPs. In contrast, all but 4 events at Monitoring Point 5 were greater than the Willamette River TDS guidance value of 100 mg/L. This is not unexpected, as Carli Creek is groundwater-fed, and likely contributes considerable TDS due solely to groundwater-derived ions (particularly during extended low-precipitation dry periods, when storm runoff is absent). Study-wide medians at each Monitoring Point appear to show small decreases in concentrations, and an actual increase within the creek for TDS/TS.

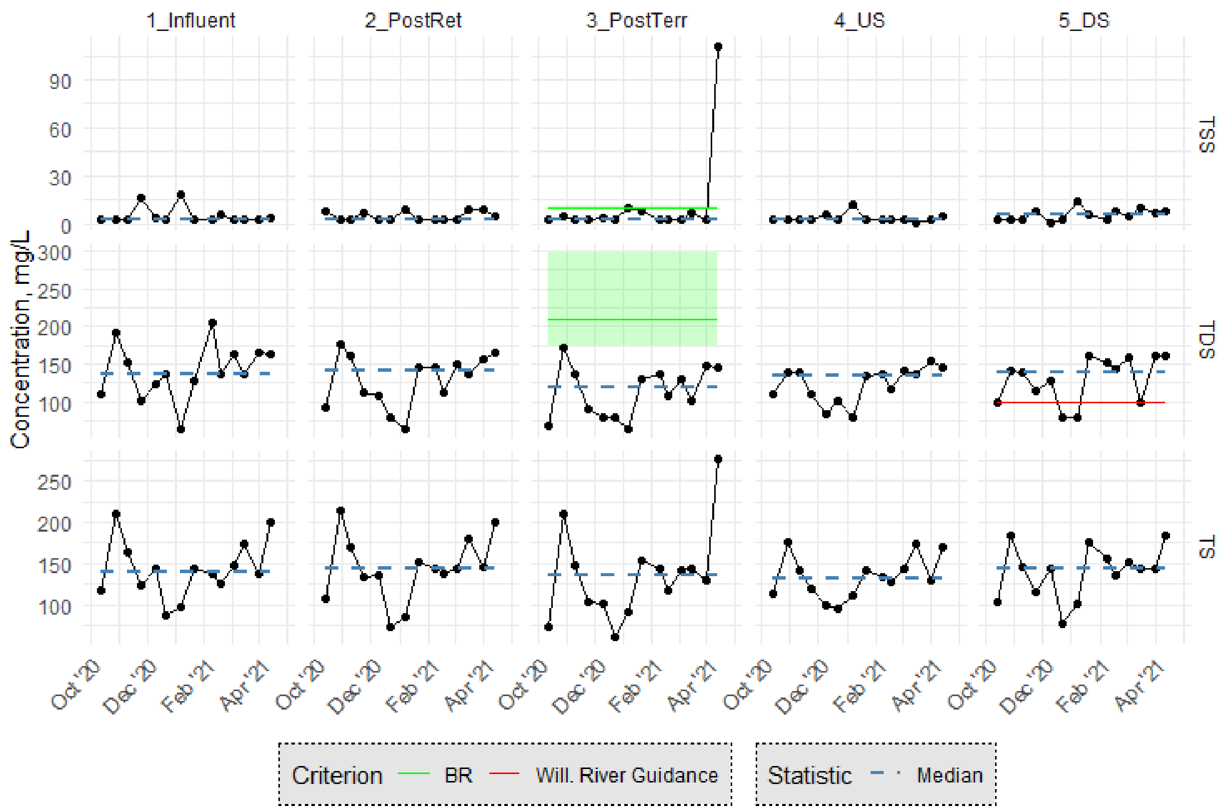

Figure 20. Total Suspended Solids (TSS), Total Dissolved Solids (TDS), and Total Solids (TS) concentrations, divided by monitoring point, compared to the Willamette River guidance values (OAR 340-041-0345(2)) and study-wide median concentrations for each parameter (dashed blue). Solid green lines represent ISWBMPdb median concentrations for bioretention BMPs, with the ribbon representing 95% confidence intervals.

Percent reductions, from a calculation and mass-basis, were also calculated to determine treatment terrace (from MP 1 to 2) treatment effectiveness to address the first research question. In the case of each pollutant, study-wide average concentration-based percent reductions were lower than mass-based percent reductions across the Terrace (Table 15). For Carli Creek, on average, each pollutant was exported (i.e., negative percent reductions) when comparing the upstream to the downstream sites. In both pair-wise comparisons, TSS had particularly high exports, although concentrations of these were consistently small (except for the last event at MP 3). Therefore, large negative percent reductions are possible with very small changes in concentrations (e.g., 2.5 to 5.0 mg/L is a 100% increase). In contrast, study-wide average mass-based percent reductions on the terrace were moderate for each fraction, due largely to the volume reductions occurring there.

Table 15. Solids mass- (n=10) and concentration-based (n=14) reduction averages and standard errors of the mean for the sampling campaign. Negative reductions imply an increase. Numbers in the column headers refer to MPs.

Parameter, Units	Percent Reductions: Terrace (from 1 to 3) Mass-based ($\bar{x}$ ± SE)	Percent Reductions: Terrace (from 1 to 3) Concentration-based ($\bar{x}$ ± SE)	Percent Reductions: Carli Creek (from 4 to 5) Concentration-based ($\bar{x}$ ± SE)
Total Suspended Solids, mg/L	18.1 ± 24	-209 ± 191	-191.8 ± 134
Total Dissolved Solids, mg/L	43.3 ± 9.9	19.2 ± 3.7	-5.7 ± 5.3
Total Solids, mg/L	35.1 ± 11.8	8.1 ± 5.0	-4.7 ± 4.4

Event-specific reductions on the terrace varied widely for these different solids fractions as well (Figure 21). TS and TDS concentration-based reductions were nearly always positive, although there was no clear pattern with the precipitation record during the study period. The largest TSS concentration-based export occurred during the large mid-January rain event. Mass-based reductions on the other hand were much higher for all solids fractions, except again for TSS. One export event coincided with the mid-January rain event, while the second occurred during the 12th event, when there was a net export of flow during the 24-hour period.

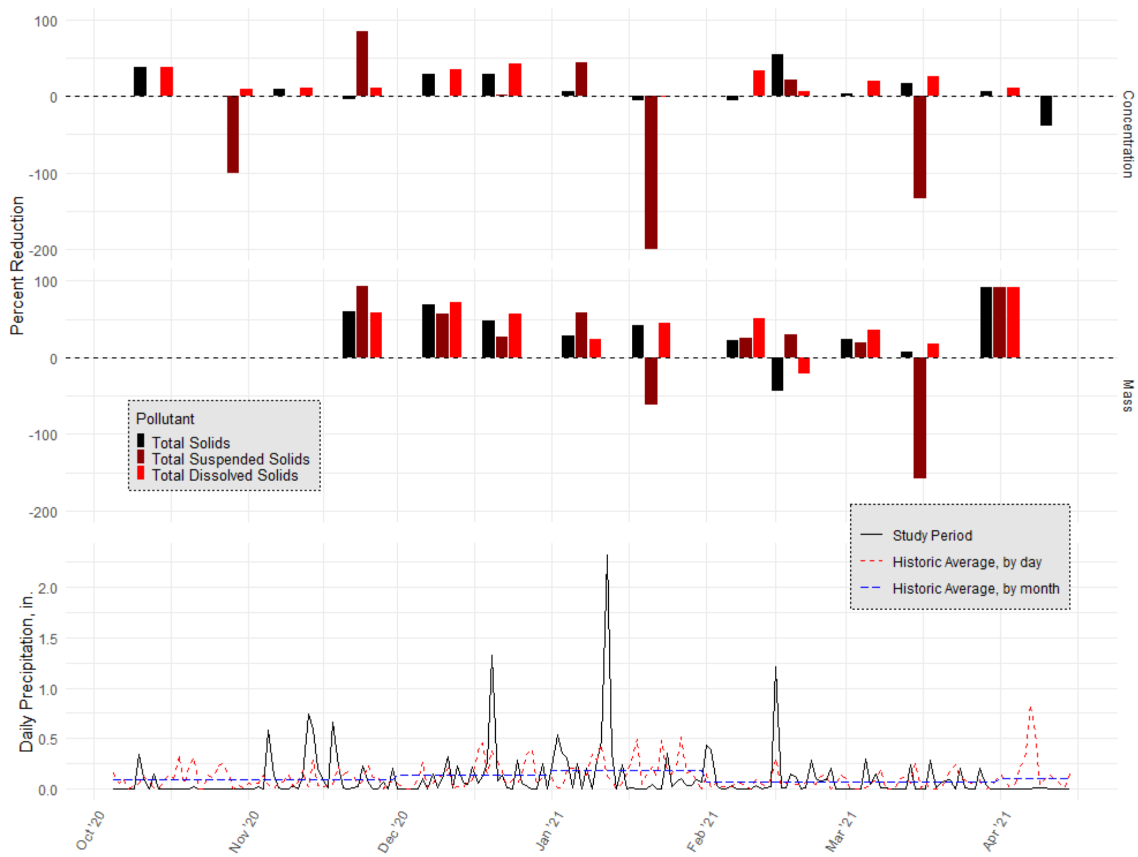

Figure 21. Mass and Concentration-based reductions for solids between Site 1 and Site 3 of the CW compared to a nearby precipitation gage across the study period. Historic precipitation record from September 2017-September 2020. Positive Reductions imply the mass or concentration decreased across the CW (i.e., from Site 1 to Site 3). Negative Reductions imply an increase.

E. coli

Bacterial indicator organisms, as *E. coli*, were measured throughout the study period at all five sites. A statistical summary is given in Table 16 below. The analytical method for *E. coli* used for this study occasionally resulted in "greater than" values due to inadequate sample collected to perform dilutions. Similar to the solids category, some values were also "less than" the detection limit of 1 MPN/100 mL. When "greater-" or "less than" values were measured, they were noted in the Table. In order to calculate some statistics (e.g., geomean) the a) detection limit for "less than" results, and b) the reported maximum value for the "greater than" results were used.

Table 16. Summary statistics for *E. coli* indicator organism in the bacteria category.

Parameter, Units	Statistic	Monitoring Points				
		1	2	3	4	5
E. coli, MPN/100 mL	GM, Median	31, 22	49, 59	14, 21	32, 30	17, 16
	σ, Range	906, 1–2610	653, 2–2420	30, 1–88	636, 5–2420	31, 2-108
	# less-than, % of total	3, 21%	0	2, 14%	0	0
	# greater-than, % of total	2, 14%	1, 7%	0	1, 7%	0

Concentrations of *E. coli* were nearly universally reduced between the terrace and Carli Creek (Figure 22). Clear decreases are observed within the creek (between MPs 4 and 5), and all events were below the Oregon Water Quality Criteria (OWQC) single-sample limit of 406 MPN/100 mL, except the 1st event upstream. Based on median concentrations, the study-wide decrease on the terrace appears less apparent, although there are fewer large results. Monitoring Point 3 results were also below (or within the 95% CI) the ISWBMPdb effluent median concentration for bioretention BMPs.

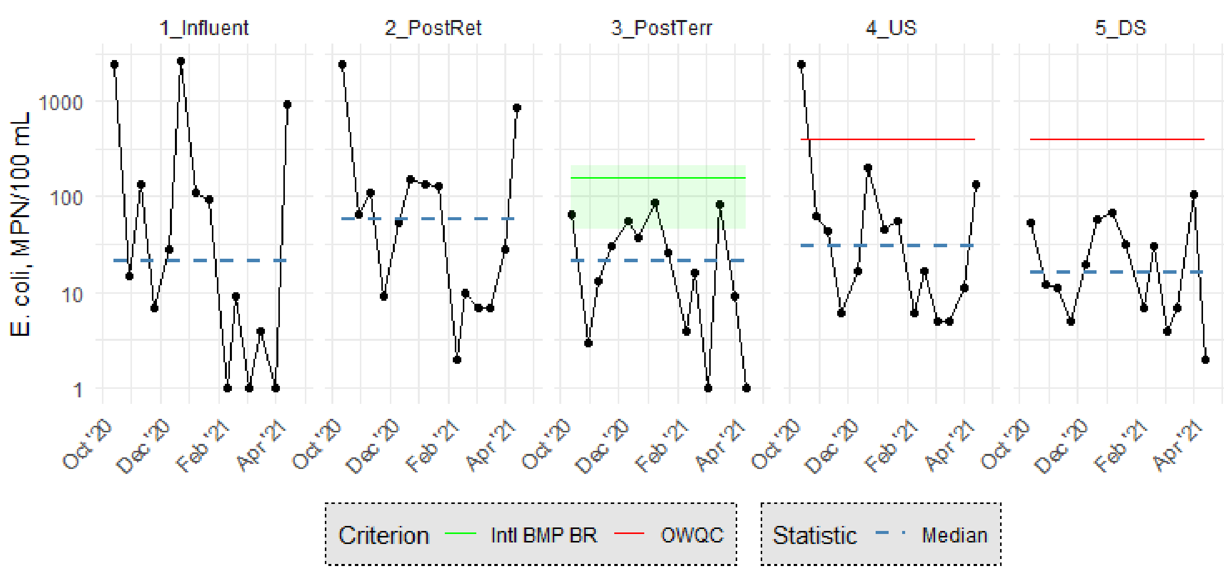

Figure 22. *E. coli* concentrations (y-scale log-transformed), by monitoring point, compared to bioretention ISWBMPdb median effluent concentrations (solid green, Intl BMP BR) and Oregon single-sample water quality criteria (solid red, OWQC), with study-wide median concentrations (dashed blue).

Another element of the freshwater contact bacteria standard is that in a 90-day period, the geometric mean of 126 shall not be exceeded (OAR 340-041-0009 (1)a). During any 90-day period of this campaign, a 90-day geometric mean of 126 MPN/100mL was never exceeded (the highest calculated geometric mean was 23 MPN/100 mL). Bacterial mass reductions were not assessed in relation to precipitation. Additionally, the log-reductions were calculated between the terrace inflow and outflow (Point 1 to 3) and the upstream and downstream sites on Carli Creek (Point 4 to 5). They ranged from -1.31 – 2.96 across the terrace and -0.99 – 1.82 across the creek. Generally, negative log reductions which correspond to *E. coli* increases corresponded to increases of very low concentrations. For example, largest increase of *E. coli* on the terrace occurred on the 12th event, and corresponded to an increase from 4 to 82 MPN/100 mL.

During the study, 64% of the events had positive *E. coli* reductions, with average log-reductions of 0.34, with the largest reductions usually occurring when terrace inflow concentrations at MP 1 were very high (e.g., during events 1, 6, and 14). In other words, the terrace was very effective at decreasing *E. coli*, sometimes up to two orders of magnitude, during this study period.

Nutrients

Nutrients measured at the terrace inflow Monitoring Point were within the same range as those measured at the historic piped location upstream, located within the MS4 system (Table 17). Ammonia and Total Phosphorous (“Total P”) in particular had similar median values to historic medians (Ammonia: 0.39 mg N/L versus historic 0.025 mg N/L; Total Phosphorous: 0.06 mg P/L versus historic 0.05 mg P/L). In contrast, nitrate+nitrite levels were elevated compared to the historic site (median: 0.93 mg N/L). Nitrate+nitrite however did have the most noticeable decrease in concentration both across the terrace and within the creek.

Table 17. Summary statistics for nutrients.

Parameter, Units	Statistic	Monitoring Points				
		1	2	3	4	5
Ammonia, mg N/L	x̄, Median	0.056, 0.039	0.053, 0.056	0.060, 0.054	0.045, 0.027	0.044, 0.033
	σ, Range	0.042 0.005–0.148	0.022, 0.005–0.093	0.038, 0.021–0.151	0.046, 0.005–0.158	0.025, 0.014–0.097
	# non-detects, % of total	1, 7%	1, 7%	0	4, 29%	3, 21%
Nitrate+ Nitrite, mg N/L	x̄, Median	1.26, 1.2	1.16,1.05	0.891, 0.767	1.12, 0.994	0.858, 0.847
	σ, Range	0.34, 0.696–2.1	0.36, 0.693–2	0.47, 0.25–2	0.34, 0.544–1.7	0.40, 0.35–1.7
Total Phosphorous mg P/L	x̄, Median	0.063, 0.06	0.074, 0.075	0.094, 0.07	0.037, 0.035	0.060, 0.06
	σ, Range	0.021, 0.005–0.095	0.014, 0.05–0.1	0.082, 0.05–0.37	0.011, 0.023–0.06	0.022, 0.01–0.11
	# non-detects, % of total	1, 7%	0	0	0	1, 7%

When evaluating concentrations of nutrients across the terrace, study-wide medians appear to show increases of ammonia, decreases of nitrate+nitrite, and little change in Total P (Figure 23). Comparisons of concentrations at the terrace outflow with ISWBMPdb effluent median concentrations for bioretention BMPs show a mixed story. For ammonia, approximately 36% of the events were above the bioretention BMP median, 36% below, and the balance within. Logically, the terrace outflow median concentration closely mirrored the bioretention BMP median. Nitrate+nitrite, on the other hand, was over the bioretention BMP 79% of the time. Total P was unique in that nearly all of the events had concentrations below the bioretention BMP, with the exception of the last event. This is likely linked to the large TSS concentration at the site during the last event.

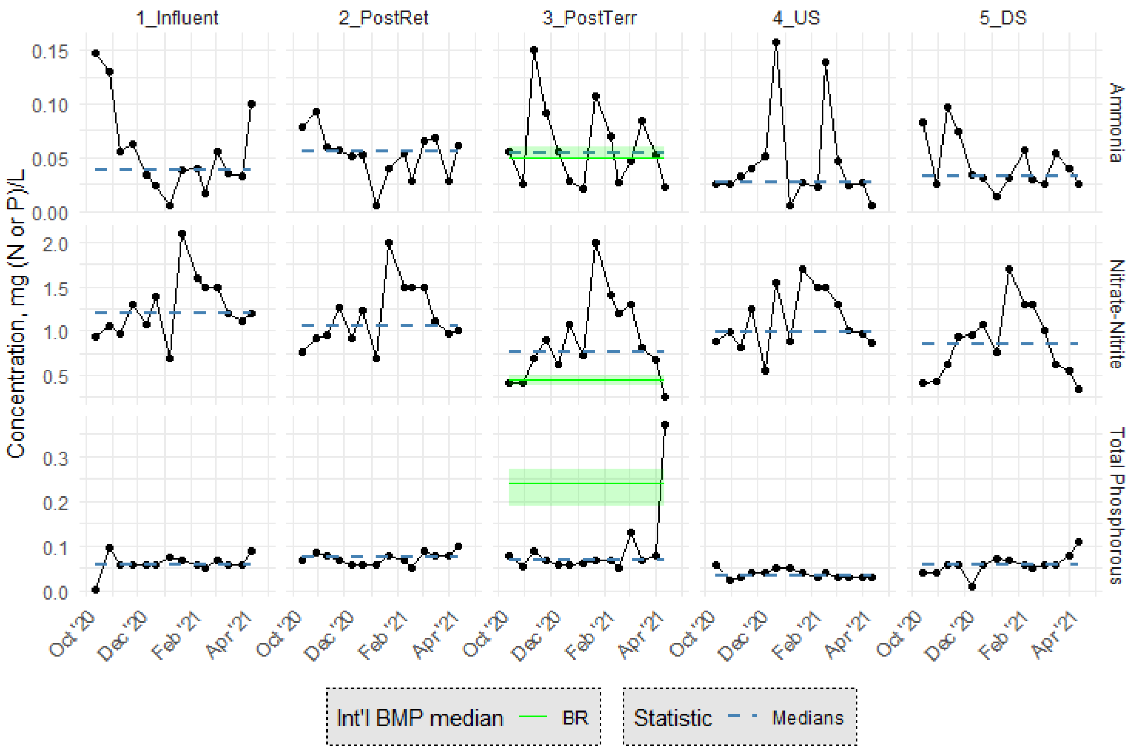

Figure 23. Ammonia-Nitrogen, Nitrate+Nitrite, and Total Phosphorous (TS) concentrations, divided by monitoring point, compared ISWBMPdb median concentrations for bioretention BMPs (solid green line; ribbon representing 95% CI) and study-wide median concentrations for each parameter (dashed blue).

Study-wide, Ammonia and Total P increased in their average concentrations across the terrace and within the creek (Table 18). Percent change on a concentration-basis for Total P across the terrace suggests high export of phosphorous, on average. However, when taking volume reductions on the terrace into account, the mass-basis percent reduction of Total P was positive, suggesting net retention. Study-wide nitrate+nitrite in the water column showed positive percent reductions from a concentration and mass-basis, with few exceptions. The fate of nitrate+nitrite is discussed below. Terrace mass-based percent reductions for nitrate+nitrite were the highest of any nutrient, in contrast to ammonia, which had the lowest (i.e., exporting ammonia), potentially indicative of ammonification and anerobic reduction of organic nitrogen.

Table 18. Nutrient mass (n=10) and concentration-based (n=14) reduction averages and standard errors of the mean for the sampling campaign. Negative reductions imply an increase. Numbers in the column headers refer to MPs.

	Percent Reductions		
	Terrace (from 1 to 3)		Carli Creek (from 4 to 5)
Parameter, Units	Mass-based ($\bar{x} \pm SE$)	Concentration-based ($\bar{x} \pm SE$)	Concentration-based ($\bar{x} \pm SE$)
Ammonia, mg N/L	-41.3 ± 32	-64.3 ± 30	-87.0 ± 38
Nitrate+Nitrite, mg N/L	41.9 ± 10	31.2 ± 6.1	22.8 ± 8.9
Total Phosphorous, mg P/L	19.4 ± 15	-140 ± 107	-74.5 ± 22

Mass-basis percent reductions were larger in magnitude for nitrate+nitrite and Total P for most events compared to concentration-basis figures (Figure 24). In contrast, ammonia was exported on 71% and 50% of the events, on a concentration- and mass-basis, respectively. For ammonia, the 10th and 12th events have mass reductions on the order of -150% and these two events coincided with the net release of water across the terrace. The largest export on a mass-basis occurred in early-January for ammonia, and was preceded by 1.69 inches (43 mm) of rain over 6 days. Nitrate+nitrite and Total P appeared fairly insensitive to even intense precipitation events. Only one and three events released nitrate+nitrite and Total P, respectively. There was a strong storm in mid-February (1.21 in./31 mm in one day), prior to the 10th event, but otherwise, these two nutrients were reduced during the study.

Figure 24. Mass and Concentration-based reductions for nutrients between Site 1 and Site 3 of the CW compared to a nearby precipitation gage across the study period. Historic precipitation record from September 2017-September 2020. Positive Reductions imply the mass or concentration decreased across the terrace (i.e., from MP 1 to 3). Negative Reductions imply an increase.

Metals

Four metals and mercury were measured throughout the study period at all five sites. Mercury was never detected at greater than the detection limit (0.2 µg/L) except once at MP 5 on the last event. Mercury analysis is expensive, and future work investigating potential mercury export in this constructed wetland should consider a) more sensitive analytical methods, b) a sampling design incorporating more specific critical environmental conditions (e.g., storm-targeted), or c) monitoring a subset of MPs on the Project. Both total and dissolved cadmium were also frequently reported at less than the detection limit (0.020 µg/L), with MP 5 tallying the highest count of non-detects for total and dissolved cadmium.

Comparisons of copper, lead, and zinc concentrations at MP 1 with historic data upstream reveal elevated total fraction values for all three metals. In other words, mean and median total copper, lead, and zinc concentrations are lower at MP 1 during this study period than historic data upstream. However, flows at MP 1 now consist of a separate mix of contributing land-uses and volumes due to the conveyance modifications made in the catchment as part of the Project (see Figures 7 and 8 above). Although the CDS unit upstream of MP 1 plays a role in reducing solids-associated metals, it's inconclusive to claim one particular cause of this deviation. Dissolved fractions are also lower, but to varying degrees. For example, dissolved lead is an order of magnitude less (historic: 0.1 μg/L, study: 0.01 μg/L) while dissolved zinc is roughly half (historic: 22.0 μg/L, study: 14.2 μg/L)

For both fractions of copper, lead, and zinc, detections were common and variable (Figure 25). Looking alone at study-wide medians for each monitoring point, it would appear metal concentrations increased across the treatment terrace (with the exception of zinc), which is the opposite pattern observed when evaluating mass-based reductions (i.e., mass loadings generally decreased). The young age of the wetland may be responsible for concentration patterns observed in this study, but is not conclusive.

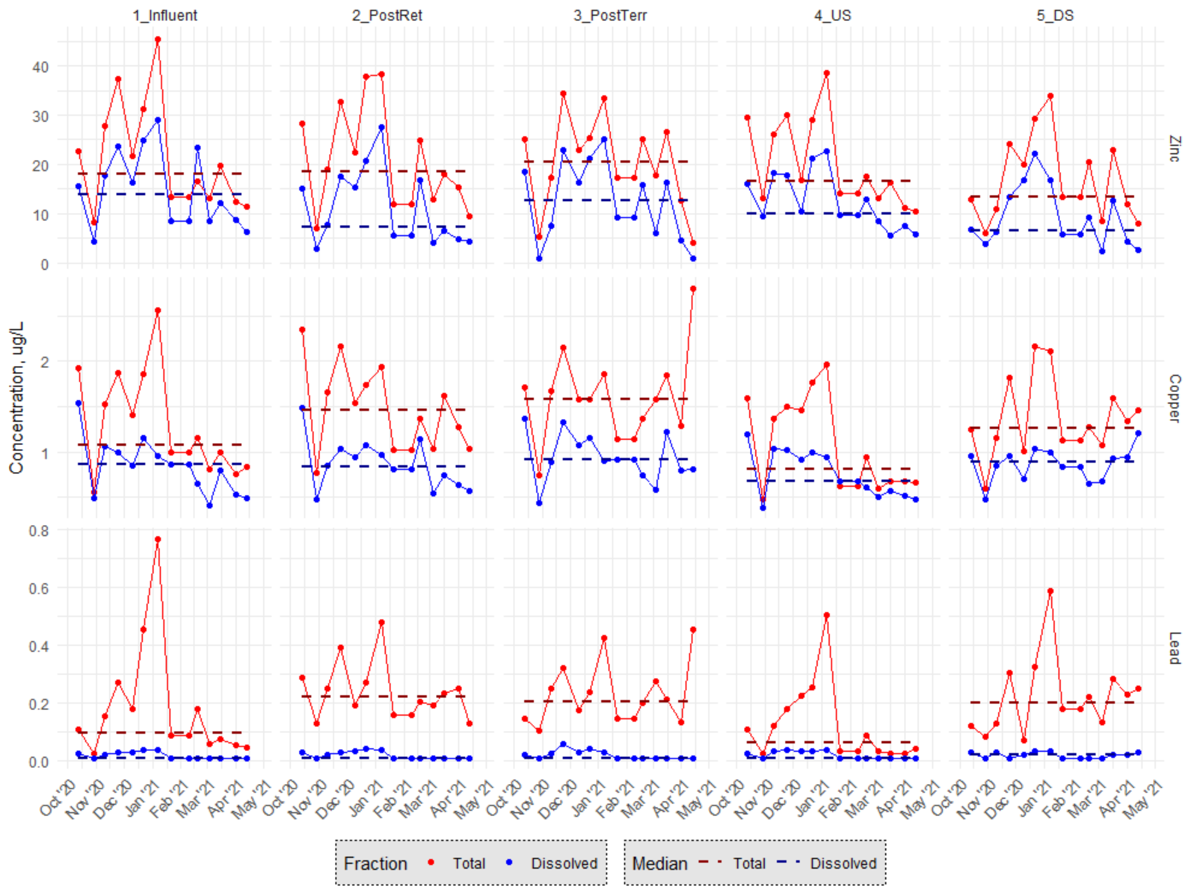

Figure 25. Total and Dissolved metals (Zinc, Copper, and Lead) concentrations, divided by monitoring point, and respective study-wide median concentrations for each metal and fraction of metal.

However, when evaluating these three metals against Oregon Water Quality Criteria, they nearly always were below the hardness- (lead and zinc) or acute/chronic Copper BLM-derived criteria (Figure 26). Concurrent hardness values were used to calculate criteria for zinc, lead, and cadmium. Even when using conservative parameter estimates (ODEQ, 2016) for the Copper BLM-derived criteria (i.e., where field collected data were unavailable), coupled with comparing the more appropriate dissolved metal fraction, only the first event exceeded in-stream state instantaneous water quality criteria ("IWQC"). When evaluating concentrations against ISWBMPdb median effluent concentrations for bioretention BMPs, copper, lead, and cadmium were all below benchmarks. Zinc median effluent concentrations were more variable, with 50% of dissolved zinc events below the median (12.5 µg/L) compared to 21% of total zinc at or below the median (12.8 µg/L)

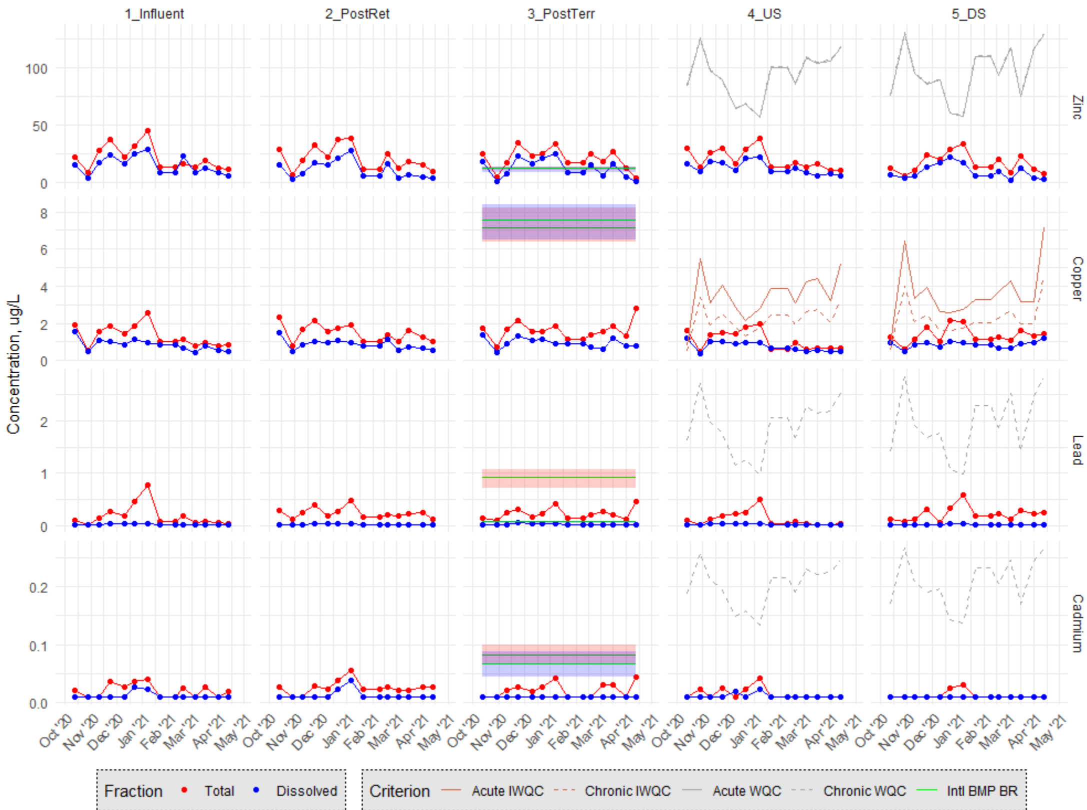

Figure 26. Total and Dissolved metals (Zinc, Copper, Lead, and Cadmium) concentrations, divided by monitoring point, compared to Oregon water quality criteria and International stormwater BMP median effluent concentrations for Bioretention BMPs.

Differences in percent reductions follow a similar general pattern as nutrients and solids in that concentration-based results suggest export of metals (with the exception of zinc) across the terrace and the creek (Table 19). Again, when accounting for volume reductions on the terrace, percent reductions are positive (except for total lead), suggesting net retention of metals. The negative mass-basis percent reduction for lead is surprising considering it is a redox-insensitive metal like zinc. Dissolved copper was reduced at the highest percent, on average, across the campaign, while total zinc was reduced the most for total fractions.

Table 19. Metals (copper, lead, and zinc) mass- (n=10) and concentration-based (n=14) reduction averages and standard errors of the mean for the sampling campaign. Negative reductions imply an increase. Numbers in the column headers refer to MPs.

Parameter (µg/L), Fraction		Percent Reductions		
		Terrace (from 1 to 3)		Carli Creek (from 4 to 5)
		Mass-based ($\bar{x}$ ± SE)	Concentration-based ($\bar{x}$ ± SE)	Concentration-based ($\bar{x}$ ± SE)
Copper	Total	4.8 ± 20	-42.7 ± 18	-36.7 ± 17
	Dissolved	43.2 ± 16	-16.7 ± 7.4	-25 ± 13
Lead	Total	-31.2 ± 39	-151 ± 63	-248 ± 82.2
	Dissolved	24.6 ± 12	-5.1 ± 6.7	-18.1 ± 17.8
Zinc	Total	13.8 ± 21	0.4 ± 9	15.2 ± 8
	Dissolved	36.6 ± 12.8	25.5 ± 10	24.0 ± 15

When assessing metal percent reductions on a concentration-basis versus rainfall, a few patterns emerge. First, all but total and dissolved zinc are generally not being reduced across the treatment terrace (Figure 27). As above, concentration-based metrics do not account for flow-reduction however, so while concentrations are necessary for comparison with Oregon Water quality criteria, they're affected by precipitation dilution effects and are not capable of integrating volume reductions occurring on the terrace. Second is that the large negative total lead value during the late October event does not occur immediately after any significant rain. A similar story is evident in the late-campaign events after March, but with other metals and fractions, such as total/dissolved copper. In other words, no large storm events precede the export of copper or lead in the final 4 events beginning in March, 2021. A contrasting pattern is generally positive percent reductions throughout the campaign for dissolved zinc, and in some cases total zinc.

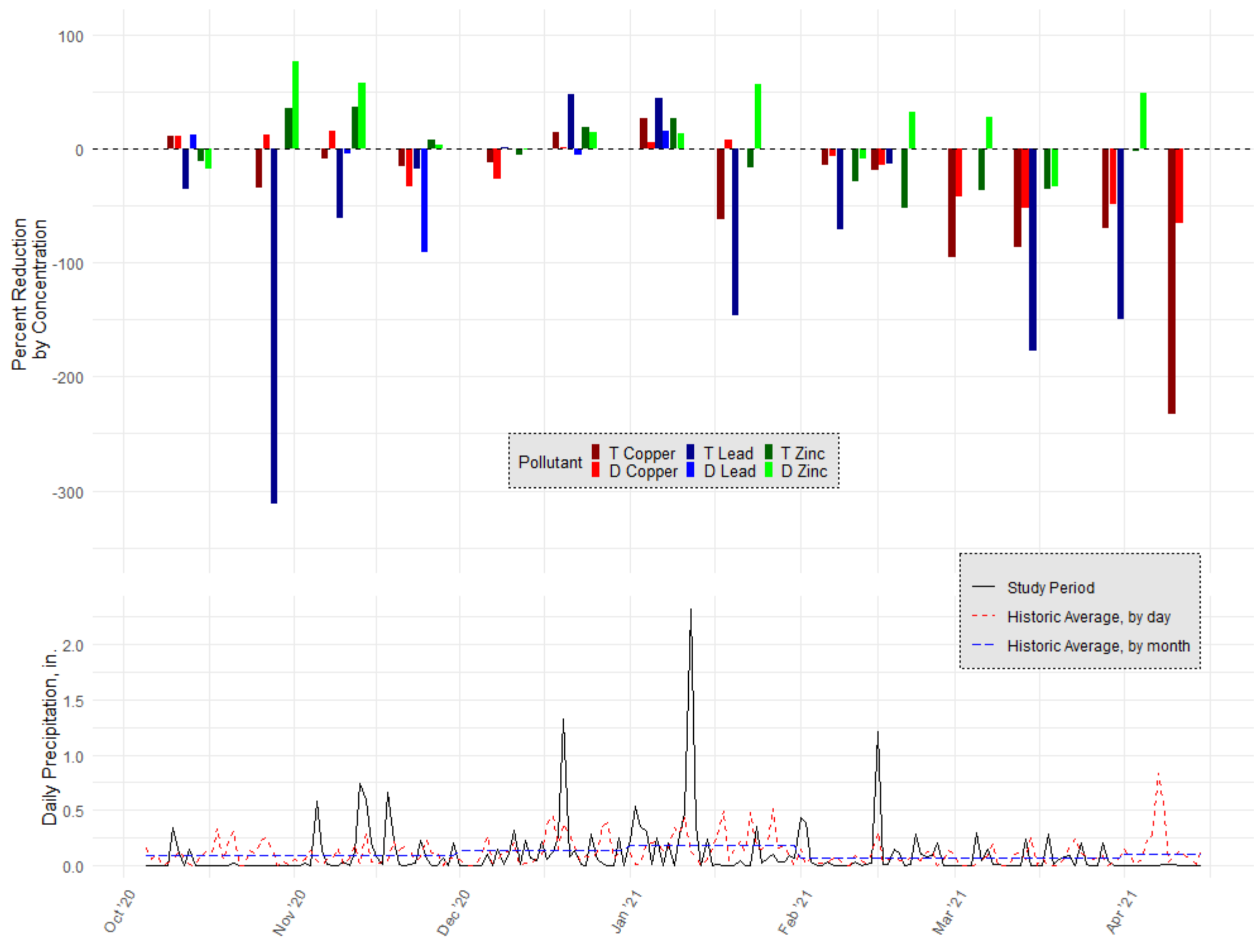

Figure 27. Concentration-based reductions for metals ("T"= total, "D"= dissolved) between Site 1 and Site 3 of the CW compared to a nearby precipitation gage across the study period. Historic precipitation record from September 2017-September 2020. Positive Reductions imply the mass or concentration decreased across the CW (i.e., from Site 1 to Site 3). Negative Reductions imply an increase.

Figure 28. Mass-based reductions for metals ("T"= total, "D"= dissolved) between Site 1 and Site 3 of the CW compared to a nearby precipitation gage across the study period. Historic precipitation record from September 2017-September 2020. Positive Reductions imply the mass or concentration decreased across the CW (i.e., from Site 1 to Site 3). Negative Reductions imply an increase.

A similarly timed trigger (i.e., in mid-February) is apparent in the mass-based percent reductions for these metals, where a sudden swing of total lead and copper (Figure 28) changes from being retained to being exported. The mid-February event, in contrast, is immediately preceded by a precipitation event. Mass-based reductions for all 3 metals do not recover to pre-storm levels until about 1.5 months later. The exceptionally well-reduced dissolved copper across the campaign is remarkable, potentially due to extensive filtering capabilities of the terrace vegetation even at this early of an age. Lead percent reductions here are illustrative of how expressing percentage changes from one very small number to a slightly larger number can appear to represent large export masses, when the magnitude of lead mass export is not that severe. For lead in particular, because the mass percent reductions incorporate both inflow/outflow concentrations *and* volume, low initial concentrations of lead and marginally higher outflow concentrations can give the impression of large export masses. For example, mass-based percent reductions for total lead on the 11^{th} and 12^{th} events were -273% and -200%, respectively, indicating export. However, this only corresponded to a net mass export of 421 g and 208 g of lead for the respective 24-hour period.

Discussion

The project objectives for this study were to answer two questions: 1) how well does the wetland reduce pollutants on a concentration and mass-basis, and 2) Do weather-related or CW-specific variables explain varying treatment effectiveness. This study answers the first question by studying 4 categories of pollutants known or suspected to be found in the stormwater runoff from this catchment and comparing concentrations with Oregon Water Quality criteria, ISWBMPdb median effluent concentrations for bioretention BMPs, and historical or literature performance. The second questions question is addressed qualitatively, in conjunction with answers from the first, by exploring pollutant removals on a concentration and mass-basis versus precipitation during the study period.

Measuring constructed wetland performance in the manner used in this study (i.e., randomized dates, reporting results based on water-quality metrics like concentration and mass) has its strengths and weaknesses. Constructed wetlands are living systems, susceptible to background levels of pollutants (e.g., nitrogen and phosphorous), stochastic variability, and seasonal affects (Kadlec and Knight, 2000). For example, long-term annual average concentrations of ammonia nitrogen in five, lightly-loaded surface-flow wetlands in the United States were at above both the terrace inflow and upstream creek MP averages for this study (North American Treatment Wetland Database [NADB], 1993). Discussed in more detail below, this suggests background loading of ammonia at the Carli Creek CW already are low, and the

CW acted as a source during this study period. Regarding seasonal affects, this study chose random dates to evaluate overall status and trends in water quality and performance when precipitation, a suspected driver of pollutant transport during the wet season, would likely stress the system's ability to reduce and retain pollutants.

This random sampling is important when assessing overall performance status and trends because conditions between critical periods (e.g., during intense rainfall) are the most common. In other words, this CW experiences rain events infrequently, and assessing performance when it is responding to these environmental stressors paints a more comprehensive picture of pollutant dynamics. However, targeting the non-growing season (i.e., cold and wet climate conditions) bakes into the study design these environmental variables. The growing season was intentionally avoided as this time of year typically had fewer and less intense storms. Cumulatively, these weaknesses in study design allow for conclusions being drawn that may leave unexplained annual pollutant reduction performance and miss the mark on describing critical periods, such as during storm-scale precipitation, even though this study was not designed investigate those critical conditions.

Hydraulic behavior, or flows throughout the wetland, was important to understand for understanding loading, but also was helpful in estimating an intentional feature of the Project. Namely, the ability of the CW to retain and infiltrate stormflows as well as reducing peak discharges. Infiltration, in the form of volume and peak-flow reduction, within CWs has long been a desirable feature for their use in management of urban stormwater runoff (Walaszek et al., 2018). Attenuation of flashy urban runoff flows from industrialized areas can return the hydrological cycles in urban streams to more pre-developed regimes, mitigating adverse effects on geomorphological processes (Bhaskar et al., 2016) that build healthy stream habitat, and counteract the ubiquitous "urban stream syndrome".

Bacterial indicator organisms like *E. coli* are important pollutants to understand, and this constructed wetland was effective at removing *E. coli* during this study period. *E. coli* concentrations are affected by several biotic and abiotic factors (ISWBMPdb, 2022) in aquatic environments like this constructed wetland. Davies and Bavour (2000) studied the fate of stormwater-associated bacteria specifically in constructed wetlands and ponds and identified the affinity of bacteria to fine suspended particles, such as clay in colloids, and suggested the fate of these associated particles are important in understanding bacteria/organism fate. In fact, sediments were shown to be reservoirs of bacteria in their study, leading to potential re-suspension and export during high flow events in similar green infrastructure such as stormwater ponds, in contrast to constructed wetlands. Low incoming suspended solids at MP 1 coupled with high mass-based dissolved solids reductions suggest that incoming *E. coli* loading is associated with dissolved or very fine (i.e., <0.45 µm) particles. Further, retention of these solids through either physicochemical processes like precipitation and adhesion or biological processes like predation are key mechanisms at play. This vegetation serves several roles including increasing detention times and increasing available surface area for biofilm adhesion (Gumbricht, 1993). Abiotic factors such as settling, sunlight, temperature, or salinity to sequester, weaken, or inactivate *E. coli* bacteria coupled with biotic forces like bactericidal compounds produced by macrophytes or predation serve to reduce bacteria concentrations.

Solids (i.e., total suspended, total dissolved, and total) appeared to be moderately reduced within the terrace on a concentration basis, but more strongly reduced on a mass-basis. With respect to solids fractions, the majority of solids measured during this study were total dissolved solids ("TDS"). The wetland was very effective at removing dissolved solids, most likely through a combination of sorption, precipitation/flocculation, and plant uptake. Dissolved solids in the sense of the analytical method used, encompasses a wide range of salts, soluble compounds (e.g., ammonia, nitrate-nitrite, and phosphorous), dissolved ions (e.g., metals), and colloids or solids capable of passing through a 0.45 μm filter. Before further discussion, it's worth noting that the likely reason for such little suspended solids observed at MP 1 of the terrace is the installation of the CDS unit, as has been mentioned before. Continued maintenance (e.g., periodic sump pumping) of this system will be important to minimize bulk, course solids being flushed through, or reversibly-sorbed dissolved pollutants from being desorbed and released into the wetland (Behbahani et al., 2021).

This predominance of dissolved solids introduced and reduced on both a concentration and mass-basis in this CW is a fortunate trait, as dissolved pollutants and their mechanisms for removal in green infrastructure are only beginning to be studied as intensively as particle-associated pollutants (LeFevre et al., 2015). Concentration-based reductions of TDS on the terrace were generally lower than those in the literature (i.e., 19% versus >50%), although one study (Merriman & Hunt, 2014; Lenhart & Hunt, 2011) did have comparable solids reductions. On a mass-basis, TDS reductions were consistent with performance data of the literature reviewed in this study. For example, of the 10 events where TDS mass-reductions were calculated, 9 of the events had positive reductions, averaging 50.5% (or 43.3% for all 10 events), compared to a range of 49-97%, a median of 50%, and an average -8%.

Assessing the solids reductions against the precipitation record during the study period does not completely explain why performance was better at certain times than others. One universal pattern is that mass-based reductions typically were higher than concentration-based reductions. Loading calculations incorporate a volume component. The percent mass reduction formula used in this study, therefore, incorporates any volume reduction (e.g., via infiltration or evapotranspiration) into the metric. The most straightforward explanation for this pattern is these volume reductions that occur on the terrace, and is a pattern observed for all of the other pollutants studied.

Nutrients, in the form of ammonia, nitrate+nitrite, and total phosphorous, showed mixed results in terms of their reductions. Total phosphorous concentrations and mass were low at the terrace inflow (i.e., MP 1). In terms of concentration and mass-reductions alone, it does appear though that Phosphorous was not reduced well, and actually was exported on a concentration basis. However, median total phosphorous concentrations were 0.06 mg P/L at MP 1, suggesting that during this study, very little phosphorous was introduced, and the terrace as well as the creek were acting as sources, not sinks, of phosphorous. Phosphorous dynamics and mechanisms of reduction are well studied (Kadlec et al., 2000) and suggest that plant senescence or remobilization of particulate-bound phosphorous is likely. Researchers (Erickson et al., 2012) have found that added iron-amended sand can help sequester dissolved phosphates, specifically dissolved phosphorous. Approaches chosen should consider the actual mechanism of phosphorous export, the phosphorous fraction, and relevant costs associated with implementing soil amendments. For example, if plant senescence is the dominant mechanism of phosphorous export, a schedule of periodic plant harvesting and hauling off-site could be employed to minimize litterfall and subsequent leaching. Based on these results total phosphorous appears limited in this system and transports little to Carli Creek in comparison to other CWs, as shown by the bioretention BMP median effluent concentration from the ISWBMPdb being well above the concentrations at MP 3.

Only two nitrogen species were monitored during this study, limiting the conclusions possible to draw beyond answering the study questions. Nitrogen cycling is a complex process in wetlands (i.e., involves intermediate nitrogen species such as N_2O, organic nitrogen/Total

Kjeldhal nitrogen, and nitrogen gas) and the reversible and irreversible transformations of nitrogen in wetland environments have been extensively studied (Lee et al., 2009; Jahangir et al., 2016; Erler et al., 2010). The significant reductions of nitrate+nitrite (NO_x) was good news. In contrast, ammonia was consistently exported throughout the study period, on a concentration and mass-basis. Nitrate transformations are microbially mediated in several ways (Burgin and Hamilton, 2007). Further, permanent (i.e., those leading to nitrogen gas formation) reduction pathways require adequate anoxic conditions, temperature, and sufficient carbon, iron, or sulfur. While anoxic conditions were not explicitly measured during this study, the path nitrate takes depends, like other mechanisms on biotic factors like resident microbial communities or vegetation density/richness and abiotic factors such as pH, hydraulic loading, conductivity, DO, or temperature.

The mechanism of NO_x reduction and ammonia export were not specifically elucidated in this study, however, clues from the constructed wetland's age, it's design features on the terrace, and historic land-use suggest certain mechanisms. For example, nitrate is likely not limited as substantial nitrate+nitrite is transported to the CW (from MP 1) and historic agricultural use of the CW area could provide a reservoir of mobile nitrogen in the soil. Some evidence does suggest that flooding of former farmland has significant short-term effects on the microbial community in the soil and N/P exports, at least in estuarine systems (Kristensen et al., 2020, Rubin et al., Marcelo et al., 2010). Also, low temperatures slow ammonia nitrification, and the NO_x reductions could be due to infiltration (nitrate is very mobile), denitrification, or plant uptake within the CW, but denitrification is likely dominant. This is because the study period occurred during the cold, non-growing season and macrophytes had had a couple years to develop colonized rhizospheres capable of supporting denitrifying heterotrophic bacteria with abundant organic carbon available.

Permanent (i.e., through nitrogen gas formation) NO_x removal can be sulfur-driven, iron-driven or through microbial respiration, depending on available oxygen and carbon sources. This irreversible process is a trademark of anoxic soils in wetland environments and will likely become more important as Carli Creek CW plants mature, hydric soils further develop, and the CW ages. For example, nitrogen removal efficiencies in the literature show mixed trends (Table 20). A Swedish CW (Al-Rubaei et al., 2016), show increases in percent removal of total nitrogen (TN) with wetland age, although the CW in question had had a nearly order of magnitude larger CW:catchment area-ratio (i.e., 2% versus 0.3%) than the Carli project, received runoff laden with salt, and was constructed in an even colder climate (Semadeni-Davies, 2006) than Oregon. On the other hand, Merriman and Hunt (2014) observed decreased NO_x removal on a mass basis over time, although concentration-based NO_x removal increased from 9 to 41 %. This could be caused by the authors' study period spanning a full calendar year and their targeting of storm events (i.e., higher N concentrations in the event inflows). The CW in their study also received no maintenance during their study period, causing a loss of hydraulic capacity via sedimentation and associated loss of event volume reductions, which could explain the decrease in % mass removal.

Table 20. Average percent mass removal efficiencies of various nutrients in other studies.

Study	CW age, yrs	Average % mass removal
Semadeni-Davies, 2006; Al-Rubaei et al., 2016	3	TN: 41 TP: 65
	16	TN: 68 TP: 92
Lenhart & Hunt, 2011	1	NH_3: 42 NO_x: 41
Merriman & Hunt, 2014	5	NH_3: 7 NO_x: 27
Adyel et at., 2017	5	TN: 48
Heyvaert et al., 2006	5	NH_3: 59 NO_3: 59
This Study	3	NH_3: -41 ± 32 NO_x: 42 ± 10

Ammonia (or more soluble ammonium/NH_4^+ at neutral pH) could be generated through nitrate reduction, ammonification of organic nitrogen (ISWBMPdb, 2022), or dissimilatory nitrate reduction to ammonia ("DNRA"), a process favored in labile-carbon rich environments compared to respiratory denitrification (Erler et al, 201). The rate of nitrification of ammonia to nitrate also slows down in low temperatures (Al-Rubaei et al., 2017, Varma et al., 2021), particularly in surface flow CWs like the Carli Creek Project, sometimes dropping up to 40% compared to summer season removal efficiencies (Song et al., 2006). Temperature of the surface water consistently decreased from MP 1 to MP 3, sometimes up to 6.2° C, creating a possible gradient of nitrification (consumption of ammonia). Minimal nitrification rates occur between 2 – 5° C (Stark, 1996) which, while not the only pathway for ammonia removal in aquatic systems, coupled with nearly steady denitrifications rates may explain ammonia exports occurring simultaneously with nitrate+nitrite reductions. Specific residence zones and low-infiltration soils incorporated into the bioretention cells' designs most likely favor denitrification since those features develop anoxic soil conditions and appropriate microbial communities more quickly. Plant uptake however cannot be ruled out, but may require an isotope-tracer study to confirm (Rhaman et al., 2019).

Daily precipitation totals did not seem to explain any of the percent concentration or mass-based reductions of ammonia. However, acute (with or without salmonids present) and chronic Oregon ammonia water quality criteria were above ammonia concentrations at all monitoring points throughout this study (United States Environmental Protection Agency [USEPA], 2013). Certain cumulative effects of precipitation such as the duration/frequency of rain absences (i.e., event antecedent dry days) and cumulative precipitation totals can act synergistically to limit the hydraulic capacity and associated infiltration capacity of the CW (e.g., median antecedent dry days for the study events was 1 day). Al Rubaei et al., (2017) found lower volume reductions during the winter season versus the rest of year, ostensibly due to a cumulative rainfall affect which would seem to adversely affect the magnitude of mass-based reduction calculations (even though reductions of ammonia were higher on a mass-basis versus concentration-basis). Specific to ammonia, frequent inundations in cold temperatures could inhibit nitrification enough to outweigh internal organic nitrogen ammonification, resulting in net export. With respect to NO_x, it appears that it was reliably reduced except after the mid-February 2021 rain event, which happened to coincide with the historic ice storm in Portland, OR. Total nitrogen removal efficiencies generally decrease with decreasing temperature with CWs (Land et al, 2016), suggesting that this icing event seriously impacted denitrification processes as well.

This study lastly looked at several metals commonly associated with stormwater runoff and found that generally speaking, all were below Oregon Water Quality Criteria, specifically at the in-stream sites (MP 4 and 5). Conservative default values were necessary when deriving the Copper BLM IWQCs. This means that if site-specific water quality parameters were used, the derived IWQC may be higher, suggesting that the levels of copper, coupled with potential exported dissolved organic carbon in the system (Chahal, et al., 2016) are highly protective of aquatic fish health in this tributary to the Clackamas River. Total mercury was also analyzed across the project during the study but the method used may not have been sensitive enough to detect mercury in the water column (detection limit of 0.2 µg/L), suggesting future studies should use more sensitive methods (e.g., Method 1631E; detection limit of 0.0005 ug/L, USEPA, 2002) in a targeted study design to offset the increased analytical cost. For cadmium, copper, and lead, all metals were below their ISWBMPdb median effluent concentration benchmarks for bioretention BMPs. Median effluent concentration benchmarks for Zinc were roughly in the middle of the 95% confidence interval range, which makes sense as the studies used for total and dissolved zinc in the ISWBMPdb were predominantly transportation-type land-uses, and is a very common pollutant around impervious industrial land-uses due to galvanized metals and tires.

Similar to total mercury, total and dissolved cadmium concentrations were very low, suggesting cadmium was not present or present at detectable concentrations in the runoff from the catchment. This is unusual considering the land-use and abundance of cadmium detections in stormwater runoff from land-use similar to the Carli Creek catchment (USEPA, 1987; Davis et al., 2001) Concentration-based reductions were mixed for the other metals except Zinc, which showed consistently fair-to-good performance in both total and dissolved fractions on the terrace and creek. In contrast, total lead showed consistent negative concentration-based reductions (i.e., exports) during the study. After the mid-February event, substantial total and dissolved copper also appeared to increase within the terrace until the end of the study. Effect patterns from precipitation were difficult to discern when evaluating concentration-based reductions, but a strong signal was observed in the mass-based reduction data after the mid-February rainfall/ice storm event. After that event, there appeared to a be a sudden shock, causing exports of all three metals, which in the case of total fractions of copper and lead, lasted for several more weeks.

Mechanisms of metal transport, transformation, and fate in green infrastructure or CWs receives substantial attention in the literature (in field studies: Knox et al., 2010, Beck & Birch, 2011, ; in mecocosm/pilot studies: Soberg et al., 2019, Lange et al, 2020, Ventura et al., 2021, Schück & Greger, 2020, Rangsivek & Jekel, 2005l in reviews: LeFevre et al., 2015, ISWBMPdb, 2022, Müller et al., 2020, Zhang et al., 2012). Metal chemistry is complex and determined by abiotic factors such as pH, DO, oxidation-reduction potential, temperature, and conductivity in addition to biologically-mediated processes (Girts et al., 1987). This study also differentiated between total and dissolved fractions. To better understand different behavior between metals, it helps to understand each metals propensity to adsorb onto solids and particulates, be oxidized, be reduced (i.e., gain electrons), bioaccumulate in wetland macrophytes, and co-precipitate/complex with other dissolved materials.

In order of adsorbing to particulate particles, zinc is the "stickiest", followed by lead, then copper. This influences how these metals partition in stormwater and, once introduced to the CW, between sediment/macrophyte compartments and the water column. Zinc was the highest-reduced metal, and reductions decreased only after the mid-February rain/ice event. This suggests that physical (i.e., settling) and physicochemical (e.g., sorption or precipitation with iron/sulfur) processes governed total and dissolved zinc fate, respectively, during this study period in the CW. Particle-bound zinc benefits from being retained in wetland soils through stormwater residence on the terrace and settling processes. Dissolved zinc fate is more complicated but based on this data, was removed as well as total zinc. Similar to Zinc, lead is also relatively "sticky" and redox-insensitive (ISWBMP, 2022), or resistant to changes in solubility based on environmental oxidizing/reducing conditions (i.e., dissolved oxygen concentration). Unexpectedly, lead was observed being exported significantly more than zinc, despite their similar characteristics. This could be a consequence of methodology with lead. In other words, using a percent change metric with very small concentrations exaggerates magnitude.

Total and dissolved copper stands apart from the other studied metals as it is a relatively soluble metal (redox-sensitive), and the least sticky of all three metals. Copper's solubility is a function of pH and temperature like lead and zinc, but is unique in its tendency to adsorb to active sites on organic materials (e.g., dissolved organic carbon, humic materials, etc., Minton, 2005). Dissolved organic carbon was not measured in this study, but is likely highest during the rainy season as exports from urbanized catchments (Kalev et al., 2021, Kalev and Toor, 2020) plants senesce, die, and decompose, releasing organic carbon. This organic material potentially aids in binding dissolved copper, facilitating precipitation or settling processes with other colloidal or fine particles. The relatively high proportion of dissolved copper indicates that non-physical (i.e., physicochemical) processes are important in reducing this metal and retaining it within the CW.

Conclusion and Recommendations

The water quality and quantity data in this study has allowed for a detailed baseline to be drawn for the Carli Creek Constructed Wetland Project. It has also allowed us to answer the two research questions that guided this work. With respect to the first question, concentration-basis reductions were poor to fair for most pollutants measured during this study, with the exception of *E. coli*, total dissolved solids, nitrate+nitrite, and total and dissolved zinc. Concentrations are "what the aquatic life experience," and what Oregon Water Quality Criteria set as the metric to achieve. Therefore, further attention is warranted to ensure continued attainment of these criteria as the CW ages. These pollutants were effectively reduced on the terrace however, and in some cases, in Carli Creek itself, although there was high variability in the magnitude of the reduction. With respect to mass, nearly all pollutants were reduced, some substantially (e.g., on average 41.9 ± 10%, 36.6 ± 13%, and 43.2 ± 16% for nitrate+nitrite, dissolved lead, and dissolved copper, respectively) due in large part to volume reduction occurring on the treatment terrace. Ammonia, on a mass-basis, was exported from the system and further study is warranted to understand the nitrogen dynamics in the CW. Maintaining important mass-reduction mechanisms long term, such as volume reductions, will ensure continued loading reductions of these stormwater pollutants to Carli Creek and the Clackamas River.

This study showed certain nutrients and metals had erratic reductions during specific events. This only sparingly appeared to be explained by total daily precipitation. With respect to the second question, precipitation was useful in qualitatively understanding concentration and mass-based reductions on the terrace, but not definitive. Stronger responses may have been apparent in relationship to precipitation if storms were explicitly targeted, as the concept of storm-transported pollutants is well-established (NRC, 2008). This study however did reveal important environmental variables worthy of further study in the future, as well as generate additional research questions. To name a few, hydraulic residence time on the terrace, available storage volume, and detailed vegetation dynamics all play a role in explaining the dominant mechanisms controlling stormwater treatment. Questions left unanswered in this study that could further Clackamas WES' understanding of this system include:

1. What other environmental variables could be driving pollutant reductions on the treatment terrace? Species richness or density? Hydraulic loading?
2. What are the dominant mechanisms on the Terrace that reduce pollutants? Does a spatial gradient of reductions exist?
3. What temporal differences might exist (e.g., during the growing season)?

As the constructed wetland continues its dynamic growth, I recommend Clackamas WES consider the following ideas and concepts to maintain this one-of-a-kind project for citizens to enjoy as much as I have while studying it.

1. All natural systems require periodic maintenance. Shortly after the Project was commissioned, the designing firm that devised important stormwater treatment elements of this system provided Clackamas WES an Operations and Maintenance manual. I recommend Clackamas WES adhere to that, but recognize it is a living document and should be revisited as new information/technology becomes available. An adaptive management approach, or an "explicitly experimental approach to learning [and doing] as a way to reduce uncertainty" (Gregory et al., 2006), will be critical to the long-term success of the Project in the face of a rapidly changing future.
2. Consider a wait-and-see approach to large maintenance projects in the early 5-10 years, but understand alternative management options exist in the literature. For example,

extensive studies on vegetation selection and maintenance exist. Shuck and Greger (2020) found specific species of macrophytes (i.e., *Carex psuedocyperus and C. riparia*) excelled at phytoremediation of heavy metals. Thullen et al. (2005), identified hummocks, or submersed islands that allow shallow emergent vegetation growth but are surrounded by deeper water, as tools for hydraulic controls and wildlife goals. Of note, some systems mature just fine, or even improve, on their own with little maintenance at all (at least in the first 5-10 years), as Merriman & Hunt (2014) found when assessing water quality improvement at a 5-year-old CW in North Carolina.

3. When or if a decision is made to restore a significant treatment unit on the terrace, ensure a variety of wetland zones are maintained (Greenway & Jenkins, 2007) and their arrangement mimics the sequential "treatment train" design of the existing CW (Wong et al., 1999).
4. Vegetation management is critical on the treatment terrace. Several hypothesized mechanisms which govern nitrogen cycling or metals retention/immobilization are associated with macrophytes. However, timing of harvest and disposition of residual is important. Removal of either plant shoots or plant shoots and roots during the late autumn with the goal of reducing nutrient exports during the winter, non-growing season can backfire. Wang et al (2015) found autumnal harvest practices such as these result in decreased radial oxygen loss and associated microbial activity during the winter, which could impact nitrification and associated denitrification. Therefore, further research on the seasonal pollutant translocation dynamics and uptake of Carli Creek Project-specific macrophytes is warranted to development a potential harvest strategy.
5. The importance of further performance research cannot be underestimated. Soliciting and supporting targeted projects attempting to further characterize performance of the Carli

Creek wetland could build a more complete picture and provide further management options to Clackamas Water Environment Services as stewards of this project site.

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Appendices

Appendix A. Carli Creek Water Quality Project Quality Assurance Project Plan.

This QAPP is attached as a separate, stand-alone document.

Appendix B: Quality Assurance/Quality Control samples results.

Pollutant/Parameter	Target RPD	Field Duplicate 1		Field Duplicate 2		Equipment Blank
		Result	RPD	Result	RPD	
E. coli, MPN/100 mL	± 0.6 log	172	-0.12	11	-0.09	<1
Total Solids, mg/L	± 20	163	7.0	132	7.5	<5.0
Total Suspended Solids, mg/L	± 20	6.5	*	6.0	*	<1.0
Total Dissolved Solids, mg/L	± 20	157	8.0	149	0.7	<5.6
Hardness, mg $CaCO_3$/L	± 20	83	1.2	85	2.4	<5
Total Mercury, µg/L	± 10	<0.20	*	<0.20	*	<0.20
Total Cadmium, µg/L	± 20	<0.020	*	<0.020	*	<0.020
Dissolved Cadmium, µg/L	± 20	<0.020	*	<0.020	*	<0.020
Total Copper, µg/L	± 20	1.23	4.2	1.19	8.1	<0.10
Dissolved Copper, µg/L	± 20	0.70	17	0.78	1.3	0.11
Total Lead, µg/L	± 20	0.177	2.9	0.13	3.8	<0.020
Dissolved Lead, µg/L	± 20	<0.020	*	<0.020	*	<0.020
Total Zinc, µg/L	± 20	19.5	28	12.2	3	4.6
Dissolved Zinc, µg/L	± 20	9.2	29	5.5	20	4.1
Ammonia Nitrogen, µg/L	± 20	0.053	28	0.056	5.5	<0.01
Nitrate+Nitrite Nitrogen, µg/L	± 10	2	0.0	0.68	0.0	<0.0625
Total Phosphorous, µg/L	± 10	0.08	0.0	0.08	0.0	<0.020

Table A-1. QA/QC sample results for pollutants during the sampling campaign. Relative Percent Difference (RPD) compares results from the parent sample and the duplicate. * indicate that either one or both results were non-detect.

$$RPD = \frac{\left|Result_{parent} - Result_{duplicate}\right|}{\left(Result_{parent} + Result_{duplicate}\right)/2} * 100\%$$

$$\log\ difference = \left|log_{10}(Result_{parent})\right| - \left|log_{10}(Result_{duplicate})\right|$$

Appendix C: Site 3 and 4 Rating Curves

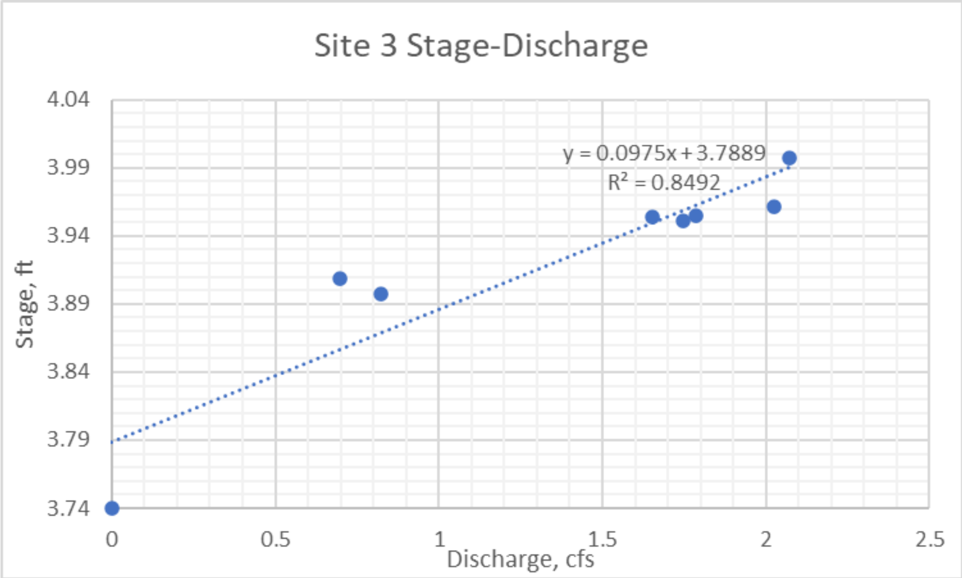

Figure A-1. Stage-Discharge Rating Curve for Site 3.

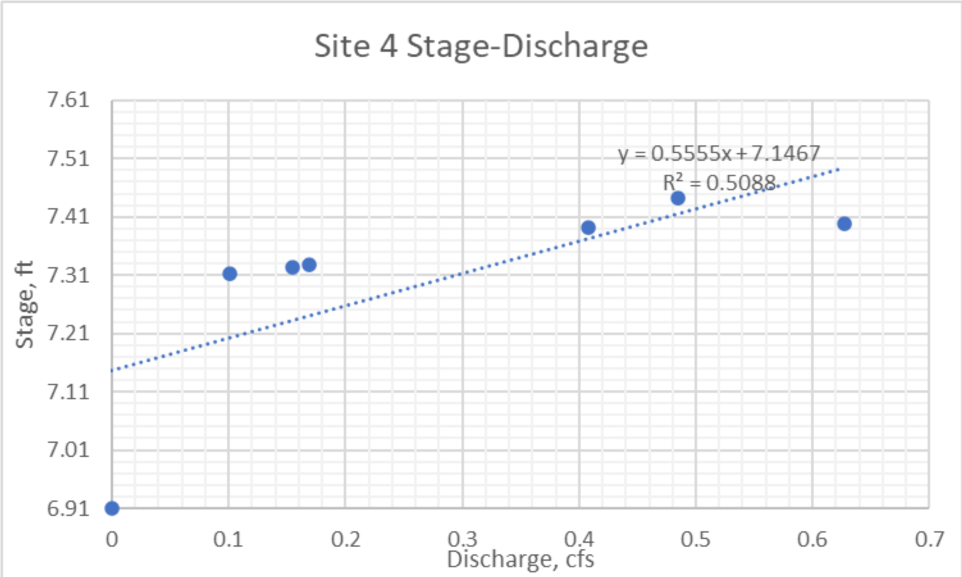

Figure A-2. Stage-Discharge Rating Curve for Site 4.

Appendix D. Metals and Mercury summary statistics.

Table 21. Summary statistics for Total and Dissolved cadmium, copper, lead, and zinc. Cadmium and Lead were often reported as less-than the detection limit; the counts of which are shown in the case of those two metals. Total Mercury was always reported below the reporting limit of 0.2 ug/L, except for one event at MP 5.

Metal (µg/L), Fraction		Statistic	Monitoring Points				
			1	2	3	4	5
Cadmium	Total	x̄, Median	0.025, 0.024	0.028, 0.028	0.023, 0.023	0.017, 0.01	0.013, 0.01
		σ, Range	0.015, 0.01–0.063	0.012, 0.01–0.056	0.012, 0.01–0.044	0.010, 0.01–0.042	0.007, 0.01–0.031
		# non-detects, % of total	5, 36%	2, 13%	5, 36%	9, 64%	12, 86%
	Dissolved	x̄, Median	0.014, 0.01	0.013, 0.01	0.01, 0.01	0.011, 0.01	0.01, 0.01
		σ, Range	0.008, 0.01–0.032	0.008, 0.01–0.039	N/A	0.004, 0.01–0.024	N/A
		# non-detects, % of total	11, 79%	12, 86%	14, 100%	13, 93%	14, 100%
Copper	Total	x̄, Median	1.32, 1.16	1.48, 1.46	1.66, 1.63	1.09, 0.92	1.36, 1.26
		σ, Range	0.57, 0.55–2.56	0.47, 0.77–2.35	0.48, 0.74–2.8	0.50, 0.48–1.96	0.43, 0.59–2.16
	Dissolved	x̄, Median	0.82, 0.83	0.85, 0.83	0.92, 0.90	0.75,0.65	0.85, 0.89
		σ, Range	0.31, 0.41–1.54	0.28, 0.48–1.48	0.28, 0.43–1.37	0.26, 0.38–1.2	0.19, 0.48–1.21
Lead	Total	x̄, Median	0.18, 0.10	0.24, 0.22	0.24, 0.22	0.12, 0.08	0.22, 0.201
		σ, Range	0.20, 0.025–0.77	0.099, 0.128–0.481	0.10, 0.103–0.456	0.13, 0.025–0.503	0.14, 0.07–0.589
		# non-detects, % of total	0	0	0	1, 7%	0
	Dissolved	x̄, Median	0.019, 0.01	0.019, 0.01	0.020, 0.01	0.020, 0.01	0.019, 0.02
		σ, Range	0.012, 0.01–0.04	0.012, 0.01–0.042	0.015, 0.01–0.059	0.013, 0.01–0.04	0.009, 0.01–0.033
		# non-detects, % of total	8, 57%	8, 57%	8, 57%	7, 50%	6, 43%
Zinc	Total	x̄, Median	21.9, 20.7	21.0, 18.5	20.7, 22.4	20.2, 17.1	16.7, 13.3
		σ, Range	10.6, 8.3–45.5	10.2, 7.1–38.4	9.04, 4.0–34.5	8.7, 10.5–38.5	8.5, 6–33.8
	Dissolved	x̄, Median	14.6, 14.2	11.1, 7.4	12.2, 12.6	12.6, 10.8	9.2, 6.6
		σ, Range	7.4, 4.3–29	7.6, 2.8–27.5	8.2, 1–25.2	5.6, 5.6–22.6	6.2, 2.3–22.1
Total Mercury		Results	all <0.2	all <0.2	all <0.2	all <0.2	0.2
		# non-detects, % of total	14, 100%	14, 100%	14, 100%	14, 100%	13, 93%