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The horizontal levee: a multi-benefit nature-based treatment system that improves water quality and protects coastal levees from the effects of sea level rise

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ABSTRACT

Municipal wastewater treatment plants in coastal areas are facing numerous challenges, including the need to provide a cost-effective approach for removing nutrients and trace organic contaminants from wastewater, as well as adapting to the effects of climate change. The horizontal levee is a multi-benefit response to these issues that consists of a sloped subsurface treatment wetland built between a coastal levee and tidal marshes. The wetland attenuates storm surges and can provide space for wetland transgression to higher elevations as sea levels rise, while simultaneously removing contaminants from treated wastewater effluent. To assess the ability of the horizontal levee to improve water quality and to identify optimal operating conditions, a 0.7-ha experimental system was studied over a two-year period. The removal of nitrate and trace organic contaminants was particularly sensitive to hydrology; rapid and near complete removal (>97%) of these contaminants was observed in water flowing through the subsurface, whereas surface flows did not exhibit measurable contaminant removal. Removal of F+ coliphage also appeared to be sensitive to hydrology, with up to 99% removal of these indicator viruses in subsurface flow. For phosphate, removal was not as sensitive to hydrology, but significant removal (>83%) was still observed when overland flow was eliminated. Although removal of contaminants did not appear to be sensitive to other design considerations, parameters such as soil texture and planting regimes affected the maximum subsurface flows, which in turn controlled contaminant mass loadings. Rapid subsurface removal of contaminants suggests that water quality benefits of these systems are limited by physical constraints (i.e., the ability of the system to maintain subsurface flow) and not chemical or biological conditions in the subsurface.

1. Introduction

Municipal wastewater treatment plants located in coastal environments are facing numerous challenges. Nutrient (i.e., nitrogen and phosphorus) discharges can impact marine and estuarine ecosystems by causing harmful algal blooms and eutrophication (Heisler et al., 2008). In addition, trace organic contaminants, such as pharmaceuticals, personal care products and household pesticides, have been detected in treated wastewater effluent at concentrations that pose risks to aquatic ecosystems (Sumpter and Johnson, 2005). Although future regulations may require additional treatment for these contaminants, retrofitting conventional treatment plants to remove them is expensive and technically challenging (Schwarzenbach et al., 2006; Foley et al., 2010). To further complicate matters, coastal wastewater treatment facilities are susceptible to flooding. As sea-level rises and the frequency of severe storms increases, wastewater treatment plants and other coastal infrastructure, as well as sensitive coastal ecosystems, will be threatened (Heberger et al., 2011). For example, in the United States, 30 cm of sea-level rise would result in flooding and loss of service at wastewater treatment plants serving more than 4 million people (Hummel et al., 2018).

The traditional approach for protecting coastal infrastructure from flooding involves the construction of seawalls and levees, at

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significant cost (Heberger et al., 2011). In 2013, a new approach for reducing the need to raise existing levees as sea-level rises, while simultaneously reducing the mass of contaminants discharged by municipal wastewater treatment plants, was designed. This system, which is referred to as the horizontal levee, consists of a sloped subsurface treatment wetland built between coastal levees and tidal marshes. The horizontal levee provides transitional wetland habitat consisting of native vegetation that protects existing levees from erosion and reduces the threat of coastal flooding by attenuating storm waves (Wamsley et al., 2010; Gedan et al., 2011; Shepard et al., 2011). Treated municipal wastewater effluent is discharged to the subsurface of these wetlands through a perforated pipe to provide water for plants growing on this elevated wedge of land. As the water flows though the subsurface wastewater-derived contaminants are attenuated. To accommodate greater applied flows, the subsurface consists of multiple layers. A surficial layer of low permeability soil (i.e., clay or loam) that is suitable for cultivating wetland plants, is underlain by coarse layers (i.e., sand and gravel) with higher hydraulic conductivities to achieve greater subsurface flows.

In natural and constructed wetlands, hydrology plays a significant role in contaminant removal. This is especially true for contaminants that are removed through microbial processes, as exemplified by nitrate. Across diverse aquatic ecosystems, variation in the proportion of nitrate removed appears to be largely explained by hydrological variables, such as residence times and water depth (Seitzinger et al., 2006). Hydraulic short-circuiting is common in constructed wetlands (Knowles et al., 2010), and can adversely impact treatment efficacy (Headley and Kadlec, 2007) by routing flows (and nitrate contained therein) around quiescent suboxic zones where denitrification activity is more significant (Seitzinger et al., 2006). For example, while denitrification walls can remove nitrate from groundwater (Schipper and Vojvodic-Vukovic, 2001; Schmidt and Clark, 2012), treatment efficiency decreases if water bypasses regions of biological activity by flowing through zones with higher hydraulic conductivity (Schipper et al., 2004). Additionally, overland flow in riparian wetlands often leads to less nitrate removal (Hill, 2000), likely due to decreased contact with plant roots and denitrifying communities in the subsurface (Willems et al., 1997). Similar effects are expected for trace organic contaminants that are susceptible to biotransformation in wetlands.

To assess the potential for using a horizontal levee to remove contaminants from treated wastewater effluent, we studied water quality and hydrological conditions over two years of operation in an experimental horizontal levee consisting of various combinations of design parameters (e.g., sediment texture, planting regimes). We monitored nutrients (i.e., nitrogen and phosphorus) and wastewater-derived trace organic contaminants because they are difficult to remove in existing treatment systems and frequently are present above concentrations of concern for aquatic ecosystems. We also monitored F+ coliphage to assess the ability of these systems to remove enteric pathogen indicators. Results from these analyses inform the design and operation of horizontal levee systems and provide a basis for assessing the performance of full-scale subsurface treatment systems.

2. Materials and methods

2.1. Field site

A 0.7-ha experimental horizontal levee was constructed in San Lorenzo, CA (37.67°N by 122.16°W) to treat a small portion (i.e., <1%) of the secondary effluent from a conventional activated sludge wastewater treatment plant operated by the Oro Loma Sanitary District (https://oroloma.org/sewage-treatment/). The treatment plant has a design capacity of 76,000 m^3 d^{-1}. The effluent was nitrified in a gravel trench upstream of the wetland system. Water quality characteristics for nitrified secondary effluent are summarized in Table S1. Native wetland plants (Section S1.3 of the SI), mainly consisting of members of the families *Cyperaceae* (sedges), *Juncaceae* (rushes) and *Salicaceae* (willows), were planted in the horizontal levee between November 2015 and February 2016. Native plants were propagated from cuttings (typically less than 3 cm) in the surficial soil approximately 15 months before nitrified wastewater effluent was introduced into the subsurface. During this period, the sloped wetland was irrigated via sprinklers using shallow groundwater from a well located approximately 50 m from the wetland. In April 2017, treated effluent from the wastewater treatment plant was first introduced to the horizontal levee at a total flow of 265 m^3 d^{-1}.

Prior to entering the horizontal levee, wastewater effluent passed through a gravel nitrification trench and a 0.8-ha surface-flow wetland planted with cattails and bulrushes (*Typhaceae* spp.) (Fig. S1 in the SI). The nitrification trench converted >90% of the ammonia in the effluent into nitrate and nitrite (i.e., average applied ammonia concentrations were 31 ± 5.1 mg N L^{-1}). Between April and November 2017, the hydraulic residence time in the surface flow wetland was approximately 11 days. Under these conditions, an average of 63% of influent nitrate was removed before entering the horizontal levee (i.e., average influent nitrate concentrations were 11 ± 4.2 mg N/L from April to October 2017). In November of 2017, the flow from the nitrification trench was rerouted directly into the influent pump station to the subsurface wetland to assure that higher concentrations of nitrate entered the horizontal levee. After November 2017, mean nitrate concentrations were 31 ± 6.3 mg N L^{-1} in the influent to the horizontal levee.

The subsurface wetland was divided into 12 parallel treatment cells, each having dimensions of 1 m deep x 12 m wide x 46 m long. The cells were hydraulically isolated from each other with clay berms and were underlain with a geotextile liner and a low permeability compacted clay layer ($K_{sat} < 10^{-6}$ cm s^{-1}). The 12 cells provided an ability to test four different wetland configurations in triplicate (Fig. S2 and Section S1 of the SI). The four treatments (i.e., swale-depression cells, wet meadows with fine or coarse topsoil, and willow/riparian cells) varied in terms of their topography, soil type and plant community.

The horizontal levee was gently sloped (1:30) and consisted of three granular media layers. From the bottom to the top, these included gravel, coarse sand and varying loam topsoil layers with hydraulic conductivities of approximately 0.25 cm s^{-1}, 0.1 cm s^{-1} and 10^{-3} cm s^{-1} respectively. The topsoil layers consisted of various mixtures of fine clay loam excavated onsite mixed with coarse sand (Section S1.2 of the SI). The topsoil layer supported plant roots and prevented rapid diffusion of oxygen into the subsurface. The higher hydraulic conductivities of the underlying layers were integrated to allow for a greater flow of water through the system. All subsurface layers were amended with organic carbon (i.e., wood chips) to promote microbial conversion of nitrate (NO_3^-) to nitrogen gas (N_2) via denitrification. Wood chips (*Sequoia sempervirens* or *Pseudotsuga menziesii*) of less than 2 cm in their greatest dimension were mixed into the sand and gravel layers by disking at 30% v/v prior to installing the loamy soil surface layer. Wood fines of less than 0.5 cm in size were mixed into the topsoil layer (10% v/v). Wood fines were composted for 12 months prior to use. See Section S1 of the SI for further details.

Nitrified treated municipal wastewater effluent was introduced into each wetland cell via perforated 5-cm diameter PVC pipes located at a depth of 5 cm below the surface within 0.6-m wide gravel trenches at the top of the slope.

Several features were incorporated into the design of the subsurface wetland to minimize hydraulic short-circuiting and to provide a means of collecting representative water samples. 0.6-m wide vertical gravel walls, oriented perpendicular to the direction of water flow, were installed in each cell at 15 and 30 m along the slope (i.e., 33 and 66% of the length of the slope). 5-cm diameter monitoring wells, screened at depths from 0.9 to 0.3 m, were installed in the center of each trench (Fig. S2 in the SI). The effluent from each cell flowed into a 0.6-m wide gravel trench at the end of the cell where it was collected in perforated 7-cm diameter pipes (located at the bottom of the trench). These pipes conveyed treated water to a monitoring well in each individual cell, where samples were collected. Water flowing out of the horizontal levee was pumped back to the headworks of the wastewater treatment plant and constituted less than 0.6% on average of the overall flow to the plant. Ultrasonic flow meters (Master Meter, Mansfield, TX, USA) collected flow data continuously in the influent piping prior to each cell and in treated water leaving monitoring wells at the southwest corner of each treatment cell.

Over the course of the 24-month study, operational parameters were varied to assess their impact on system performance (Fig. S3 in the SI). During the first phase of the study, from April to November 2017, the total applied flow setting was 265 $m^3\ d^{-1}$ (~22 $m^3\ d^{-1}$ per cell) and water flowed from the nitrification facility into the surface flow wetland before being applied to the subsurface wetland cells, as described previously. In July 2017, the applied flows going into each cell were adjusted to achieve similar levels of treatment across cells. During the second phase, between November 2017 and July 2018, the overall applied flow setting decreased to 190 $m^3\ d^{-1}$ (~16 $m^3\ d^{-1}$ per cell) and water flowed directly from the nitrification facility into the subsurface wetland cells. During the third and final phase, between July 2018 and April 2019, the flow setting decreased to 95 $m^3\ d^{-1}$ (~7.9 $m^3\ d^{-1}$ per cell). During this last phase, applied flows to individual cells were again adjusted to eliminate overland flow in the majority of cells (D-L) (details are included in Section S2 of the SI).

2.2. Sample collection

Water samples were collected on a monthly or biweekly basis starting in April 2017 (Fig. S3 in the SI) from the influent pump station and monitoring wells located at the end of the treatment cells. Additional samples were collected from the influent and effluent of the surface-flow wetland and periodically from intermediate wells in the subsurface wetland. Samples for chemical analyses were collected using a Masterflex E/S portable water sampler (Cole-Parmer, Vernon Hills, IL, USA) and analyzed in triplicate (between April 2017 and July 2018) or duplicate (between July 2018 and April 2019). At least two well volumes were purged prior to collecting a sample when appropriate. Samples were filtered on-site through 0.7-μm glass fiber filters into 50-mL polypropylene centrifuge tubes and immediately stored on ice prior to analysis, which normally occurred within 24–48 h. Samples for F+ coliphage analysis were collected in triplicate into acid-washed Nalgene bottles. Bottles were triple rinsed with sample water at the site before collecting samples. Samples were stored on ice during transport to the laboratory. Electrical conductivity and pH were measured at each sampling location in the field using an Ultrameter II (Myron L Company, Carlsbad, CA, USA). Dissolved oxygen and temperature were measured in the field with a YSI ProODO Optical Dissolved Oxygen probe (YSI Inc., Yellow Springs, OH, USA).

Porewater samples were collected at depths ranging from 0.1 to 0.9 m into Luer-Lok BD syringes using stainless steel PushPoint sediment porewater samplers (MHE Products, East Tawas, MI, USA). These samples were filtered on-site through 0.7-μm glass-fiber filters or 0.2-μm nylon filters and stored on ice prior to analysis.

2.3. Sample processing and analytical methods

Field-filtered samples were stored at 4 °C upon returning to the laboratory and were analyzed using established methods.

Within 6 h of collection, subsamples for ion chromatography analysis were filtered through 0.2-μm nylon filters into 0.5-mL PolyVials, capped with filter caps, and refrigerated prior to analysis, which normally occurred within 4–36 h of processing. Samples for cation analyses were acidified to pH < 5 to limit volatilization of ammonia prior to analysis. Inorganic anions (Cl^-, NO_2^-, Br^-, NO_3^-, PO_4^{3-} and SO_4^{2-}) and cations (Li^+, Na^+, NH_4^+, K^+, Mg^{2+} and Ca^{2+}) were measured on Dionex Aquion Ion Chromatography systems (Thermo Fisher Scientific, MA, USA). Anion measurements were performed with a Dionex IonPac AS23 column according to U.S. EPA Method 300.0 and cations measurements were performed according to previously described methods (Thomas et al., 2002) with minor modifications by using a 3.0 mM methanesulfonic acid eluent and a Dionex IonPac CS16 column.

15-mL aliquots of each field-filtered sample were transferred for TOC analysis into 24-mL borosilicate glass sample vials that had been rinsed with deionized water and baked at 450 °C for 4 h prior to use. Analysis of non-purgeable organic carbon (NPOC) and total nitrogen (TN) was performed on a Shimadzu TOC-V/CSH analyzer with an attached TN-1 unit according to standard methods (Method 5310B; APHA, 2012). Organic nitrogen concentrations were calculated by subtracting concentrations of nitrate, nitrite and ammonium from total nitrogen measurements.

Concentrations of a suite of trace organic contaminants were quantified according to previously described methods (Jasper et al., 2014a; Prasse et al., 2015; Bear et al., 2017) with minor modifications. Briefly, field-filtered samples were held at 4 °C for 24 h to allow reduced iron to oxidize and precipitate. These samples were filtered through 0.2-μm nylon filters to remove particulates, which mainly consisted of Fe(III) oxides. To assess potential losses, concentrations of trace organic compounds in filtered samples and samples acidified to approximately pH 2 with HCl were compared. Because acidification could cause artifacts or damage the HPLC/MS-MS system and no significant differences were observed in concentrations measured with the two pre-treatment methods (p-value > 0.6), the filtration method was used for all analyses. Samples were amended with a mixture containing stable isotope-labeled pharmaceuticals (5 ng of each) and analyzed using isotope dilution liquid chromatography/tandem mass spectrometry (Agilent 1200 series HPLC and Agilent 6460 triple quadrupole mass spectrometer).

F+ coliphage were enumerated using previously published methods (Sinton et al., 1996; EPA Method 1601). Briefly, magnesium chloride was added to 500 mL samples to achieve a final concentration of 0.05 M. Samples were subsequently filtered through a negatively charged 0.45-μm membrane filter (Millipore; HAWP04700) to capture the viruses. Filters were placed gridded-side-down into a 47-mm diameter plastic Petri dish containing 300 μL of sterile 1:1 glycerol:PBS solution. The petri dishes containing the filters were frozen at −80 °C until further processing (within 6 months). To enumerate the coliphage, the coliphage were eluted from the membrane filters using a solution of 3% beef extract, 3% Tween-80, and 0.3 M sodium chloride. 2 mL of the elution solution was added to each Petri dish and the dish was rocked for 5 min on a shaker table at room temperature. The elution liquid was removed from the dish and coliphage was enumerated in the liquid using EPA Method 1601 (DAL method). The filter was placed on solid tryptic soy agar (TSA) media containing the appropriate concentrations of ampicillin and streptomycin antibiotics and the *E. coli* host strain (EPA Method 1601). The numbers of PFU obtained from assaying the liquid media and present on the filter were added together to obtain the concentration of PFU in the assayed water.

3. Results and discussion

During the two-year monitoring period, the horizontal levee processed approximately 126 × 10^3 m^3 of municipal wastewater effluent. To assess water quality improvements that occurred as water passed through the system, we measured contaminants and water quality parameters in over 1000 samples of influent and effluent, as well as over 300 porewater samples. We also measured water flows along with other design and operational variables. The full dataset can be found on Mendeley Data (Cecchetti et al., 2020).

3.1. Water balance

In the experimental wetland system, inflows of municipal wastewater effluent and a small volume of precipitation (which constituted less than 1% of the total volume of water entering the system during the monitoring period and therefore did not have a significant impact on results) were balanced by outflows through the effluent pipes and evapotranspiration. Water flowed along the ground surface (i.e., overland flow), passed through the subsurface and evaporated or was transpired by the plants (Fig. 1). Over the 2-year study period, evapotranspiration accounted for the loss of approximately 25% of the water. The remaining water left the system through the outlet pipe. Of this remaining flow, the contributions of overland flow and subsurface flow varied considerably during the three phases of the study (Table 1). During phases 1 and 2, overland flow was approximately 2.5 times the magnitude of subsurface flow. During phase 3, swale-type cells continued to have high overland flows (i.e., around 5 times subsurface flows), though overland flow was negligible in most cells (i.e., wet meadow cells with both fine and coarse topsoils, and willow/riparian cells). Methods used to calculate flows are detailed in Section S3 of the SI.

3.1.1. Effect of hydrology on contaminant removal

The water balance in the horizontal levee was important due to its influence on contaminant removal. Evapotranspiration removed water from the subsurface, concentrating dissolved species in the remaining water. Subsurface and overland flows mixed together prior to the final sample collection point, but the water experienced different conditions. The very short hydraulic retention times in the overland flow led to little, if any, removal of contaminants, while nearly complete removal of contaminants was observed in the subsurface flow. Therefore, hydrological variables (e.g., the fraction of subsurface flow) largely determined contaminant removal. Negative correlations ($r^2 > 0.6$) were observed between the fraction of overland flow and the fractional removal of nitrogen species, pharmaceuticals, and F+ coliphage (Fig. 2). Using standardized multiple linear regressions, additional variables (e.g., temperature, planting regime) were shown to be less influential on removal of studied contaminants, with the exception of organic nitrogen and acyclovir (see section S5 of the SI for more details). The most significant correlation for most contaminants studied was between subsurface flow and contaminant removal.

These observed contaminant removal trends were attributable to efficient removal in the subsurface with little or no contaminant removal in the overland flow. For example, in the case of nitrate, the overland flow experienced short hydraulic residence times (approximately 0.4–1.0 days; section S4 of the SI) and aerobic conditions, whereas subsurface flow was characterized by longer hydraulic residence times (i.e., approximately 12–20 days) and anoxic conditions that are conducive to microbial denitrification. For trace organic contaminants, the lack of contact with biofilms that coat organic matter, plant roots and fluctuating redox conditions in the subsurface reduced contaminant removal in the overland flow. For viruses, we would expect a variety of mechanisms to increase the removal of F+ coliphage in the subsurface. For example, filtration facilitated by attachment to solids, virus inactivation and rhizosphere processes could all contribute to the high levels of removal observed in cells with less overland flow (Vidales et al., 2003; Muerdter et al., 2018). For phosphate, limited contact with phosphate-adsorbing mineral surfaces prevented significant removal in the overland flow. The assumption that little removal of contaminants occurred in overland flow was verified through the collection of samples from the water flowing over the wetland surface (Fig. 3a), which were consistently statistically indistinguishable ($p > 0.05$) from the influent.

Porewater samples collected deeper than 0.1 m indicate that most contaminants were removed in the subsurface within 5 m of the inlet to the horizontal levee (Fig. 3). The subsurface residence time in the first 5 m of the slope (approximately 0.5–1.0 days) was similar to residence times in the overland flow. The significant and rapid subsurface removal of contaminants was likely due to a combination of mechanisms. Diffusion of oxygen into the subsurface was limited by overlying fine sediments, preventing re-introduction of oxygen and promoting anaerobic processes. The subsurface also provided ample organic matter (i.e., decomposing woodchips, plant roots and exudates) on which microbial communities can obtain energy, promoting microbially-mediated processes such as denitrification (Kadlec and Wallace, 2009).

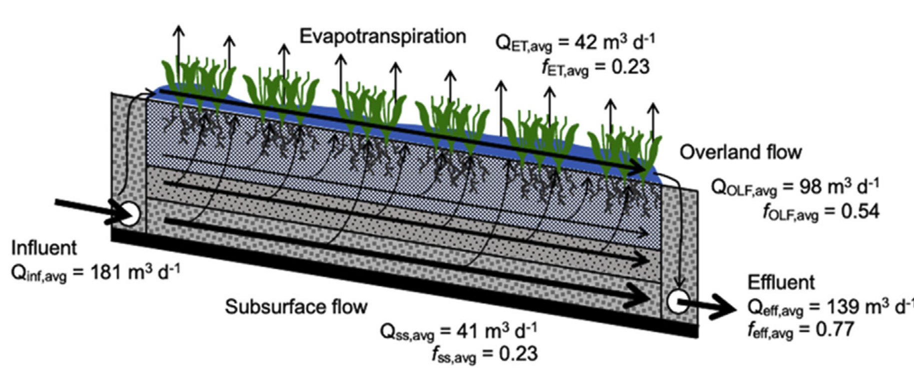

Fig. 1. Schematic representation of the water balance at the horizontal levee test facility using average values from throughout the monitoring period. Calculated values for the average magnitude of each flow (in $m^3\ d^{-1}$) are presented with the average fraction of the influent flow that each flow represents. Figure is not to scale.

Table 1
Average flow rates and fractions of the influent flow of the various flows in the test facility during operational phases 1 and 2, for all cells during phase 3 and for cells without overland flow during phase 3.

Flow Component	Phases 1 and 2 All Cells		Phase 3 All Cells[a]		Phase 3 Cells without Overland Flow[b]	
	Flow rate, $m^{-3}\ d^{-1}$	Fraction, unitless	Flow rate, $m^{-3}\ d^{-1}$	Fraction, Unitless	Flow rate, $m^{-3}\ d^{-1}$	Fraction, unitless
Influent	222	1.0	104	1.0	48	1.0
Evapotranspiration	50	0.22	35	0.34	25	0.53
Overland flow	124	0.56	40	0.38	0.6	0.01
Subsurface flow	49	0.22	30	0.28	22	0.46
Effluent	173	0.78	70	0.67	23	0.47

[a] Average sum of all cells during Phase 3, including swale-type cells.
[b] Average sum of cells during Phase 3, excluding swale-type cells but including all other cells.

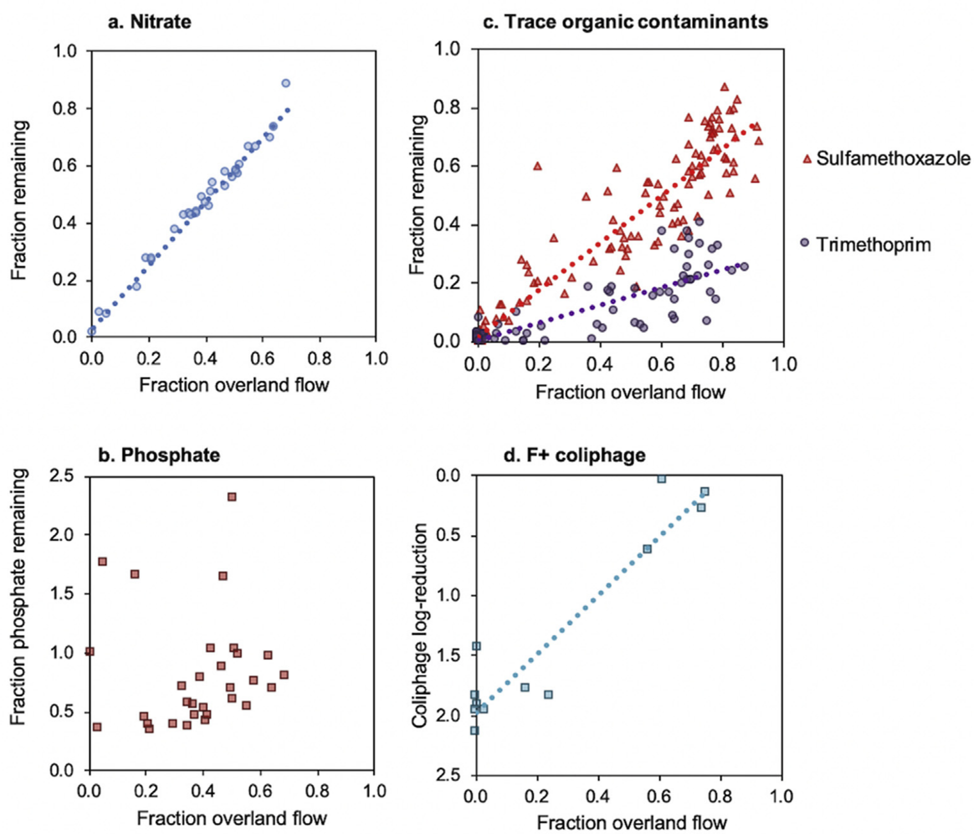

Fig. 2. The fraction remaining in the effluent of a suite of contaminants, including (a) nitrate (linear regression, $r^2 = 0.98$), (b) phosphate (linear regression not shown, $r^2 = 0.00$), and (c) pharmaceuticals (linear regressions, sulfamethoxazole: $r^2 = 0.91$; trimethoprim: $r^2 = 0.73$), and the log-reduction of (d) F+ coliphage (log-linear regression, $r^2 = 0.89$), as a function of overland flow. Values in plot (a) and (b) are flow-weighted averages across the full wetland at each time point. Values in plots (c) and (d) are data from individual wetland cells at each time point.

Consistent with these observations, removal of contaminants was most significant during the third phase of treatment when overland flow was eliminated in most cells (section S2 of the SI). During this period, over 96% of the mass of nitrogen and nitrate, and 92–99% of trace organic contaminants were removed, compared to 38–48% and 54–86% for the periods with more overland flow (i.e., phases 1 and 2).

In contrast to nitrogen species and trace organic contaminants, hydrologic conditions (e.g., percentage overland flow) did not appear to have a consistent impact on the removal of phosphate (Fig. 2b). Although removal of phosphate was high during the period when overland flow was eliminated in most cells (averaging 81 ± 23% removal), phosphate removal was poorly explained by hydrological variables in standardized multiple linear regressions, even when combined with other design and operational variables (overall $r^2 = 0.34$ for phosphate compared to $r^2 > 0.75$ for other contaminants). Removal of phosphate varied over time, ranging from 74% removal at best to concentrations more than doubling through the full horizontal levee (Fig. S5). In addition to plant uptake, phosphate exhibits an affinity for a variety of minerals (Holtan et al., 1988; Yao and Millero, 1996) and forms precipitates under certain conditions (Egger et al., 2015; Rothe et al., 2016). Biogeochemical cycling of other elements, such as iron and carbon, can have complex and variable impacts on these mechanisms in freshwater systems (Caraco et al., 1989; Murray, 1995; Szilas et al., 1998; Lin et al., 2018), which may partly explain the observed

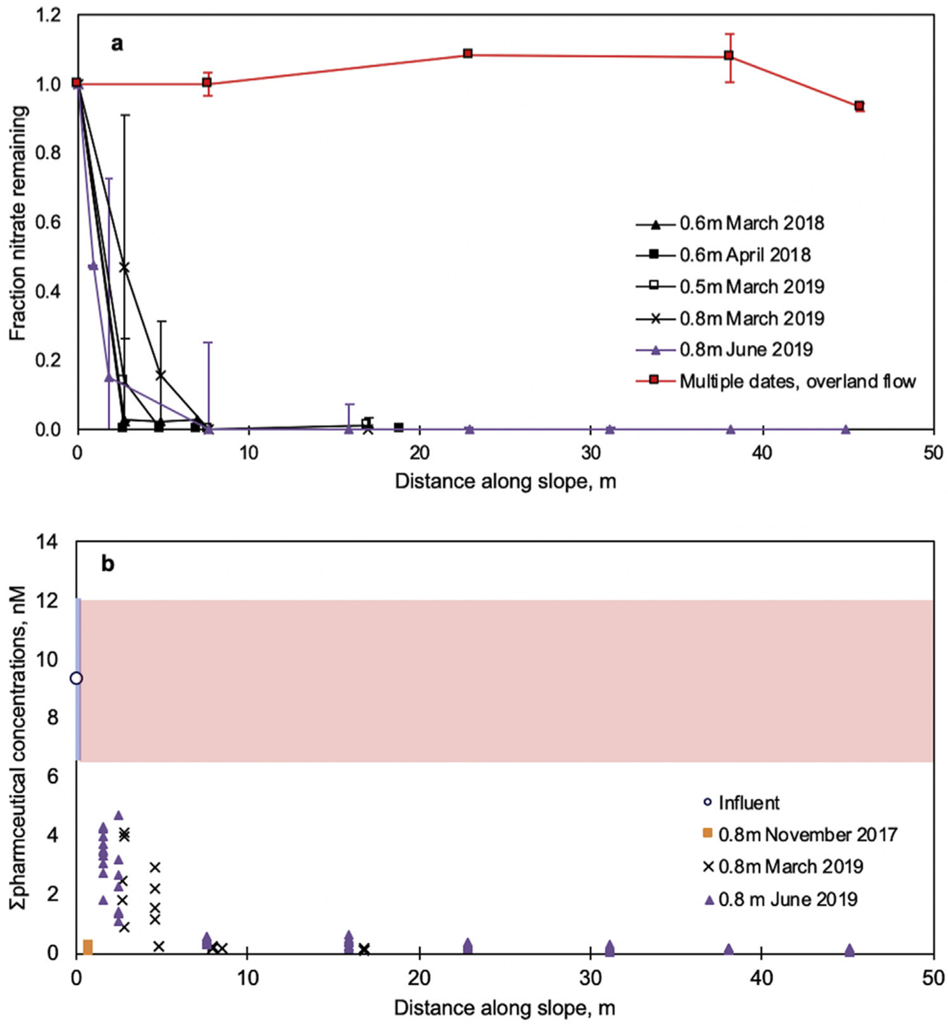

Fig. 3. (a) Fraction of influent nitrate load remaining in porewater and overland flow samples at various distances along the slope. Error bars show one standard deviation. (b) The combined concentrations of the suite of monitoring pharmaceuticals in porewater samples at various distances along the slope. A red horizontal bar denotes the range of influent pharmaceutical concentrations observed. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

variability in phosphate removal.

Regardless of the cause of this observed variability, we do not expect long-term phosphate removal in horizontal levees because these mechanisms largely involve storage mechanisms that eventually will be exhausted. For example, following plant uptake, decomposition of plant residues will eventually release a significant portion of assimilated phosphorus back into the subsurface. Additionally, even if the amount of phosphate adsorption sites on minerals in the subsurface remains constant, they will eventually be exhausted without the addition or formation of more mineral surfaces. Further research is required to understand how horizontal levees could be designed and operated to achieve substantial long-term phosphate removal.

3.2. *Effect of design and operational parameters on hydrology*

The interplay between overland flow and subsurface flow was the dominant factor controlling contaminant removal in the horizontal levee, but design and operational parameters (e.g., substrate type, plant species) also had significant impacts on the magnitudes of those flows and therefore on mass removal of contaminants. The impact of cell design on contaminant removal was unclear based on fractional contaminant removal alone because flows were adjusted in an effort to obtain similar flow distributions. There also were not significant differences in subsurface contaminant removal trends between cells based on porewater observations. However, there were significant differences in the water balance for the different cell treatment types (Fig. 4a). These differences in flows translated into significant differences in contaminant mass removal rates (Fig. 4b), suggesting that design decisions, like plant species and soil type, can have a significant impact on the treatment capacity of horizontal levees.

3.2.1. *Impacts of cell design on subsurface flows*

Over the full monitoring period, there were significant differences in flows based on soil texture. As expected, subsurface flows were significantly higher (i.e., approximately twice as high) in coarse sediment meadow cells when compared with those constructed with fine sediments (i.e., 4.6 $m^3\ d^{-1}$ versus 2.3 $m^3\ d^{-1}$; p-value < 0.001) because soil texture is correlated with hydraulic conductivity in granular media (Kadlec and Wallace, 2009). However, the magnitude of these differences decreased throughout the monitoring period. For example, during the last monitoring phase, coarse textured cells exhibited flows that were only 54% higher

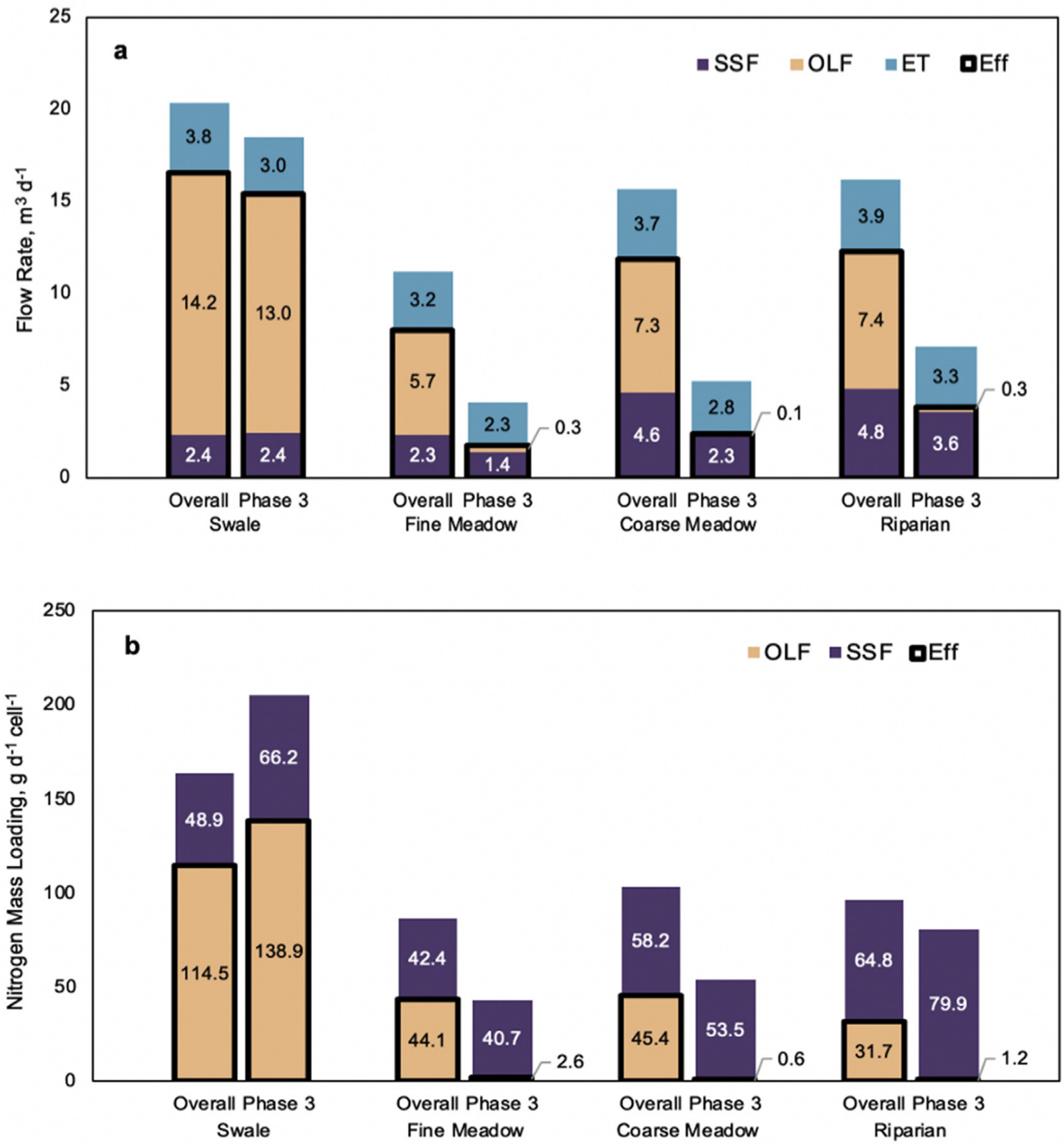

Fig. 4. (a) Average daily flow rates and (b) nitrate-N mass removed per cell type during the entire monitoring period (bars to the left for each treatment type) and during the third and final phase of the monitoring period (bars to the right). Abbreviations: SSF = subsurface flow; OLF = overland flow; ET = evapotranspiration; and Eff = Effluent flow.

than fine textured cells.

Cells planted with the willow trees, which were constructed with coarse sediments, had the highest subsurface flows and the highest contaminant mass removal rates among tested cell types throughout the full monitoring period. Subsurface flows averaged 4.9 m^3 d^{-1} in these cells, compared with flows of 4.6, 2.3 and 2.4 m^3 d^{-1} in coarse and fine wet meadow cells, and swale-type cells, respectively. Differences in subsurface flows between the willow and the coarse meadow cells were not significant (p-value = 0.92) over the full monitoring period. However, during the third phase of treatment, the differences in subsurface flows between the two coarse-cell planting regimes (willow cells and meadow cells) increased significantly. During the final phase, the willow cells had 59% higher subsurface flows than coarse meadow cells (p-value = <0.001), at 3.6 m^3 d^{-1} versus 2.3 m^3 d^{-1}, corresponding to differences in mass removal of nitrogen on the order of 10 kg N per year per cell (Fig. 4). The observed impact of riparian planting regimes on subsurface flows was consistent with past research. Willows have extensive rooting zones (Kuzovkina and Volk, 2009), which lead to greater subsurface flows in stormwater bioinfiltration systems (Read et al., 2008). In the horizontal levee, the willow cells also appeared to play an important role in increasing the volume of water that could be processed by a cell.

Although subsurface flow in swale-type cells was the lowest throughout phases 1 and 2, these cells had the second highest subsurface flow during the third monitoring phase. There was an average 40% decrease (p-value < 0.001) in the subsurface flow capacity of other cell types, likely due to a variety of clogging mechanisms (section S7 of the SI), whereas subsurface flows of swale cells did not decrease significantly (p-value = 0.85) when compared to earlier monitoring periods. The unique topography of swale-type cells, with swales running down the center of these cells (Fig. S1.2), may have played a role in the observed subsurface flow conditions (see S8 of the SI for more details).

3.2.2. Impacts of cell design on evapotranspiration and overland flow

Throughout the monitoring period, evapotranspiration rates were similar among cell types, with average evapotranspiration rates ranging from 3.2 to 3.9 m^3 d^{-1}. Only differences between fine and coarse meadow cells were significant (p-value = 0.02). However, in the third monitoring phase, more significant differences among the cells emerged, with willow cells (3.3 m^3 d^{-1}) exhibiting significantly higher evapotranspiration rates than both coarse (p-value = 0.04) and fine (p-value < 0.001) meadow cells (2.9 and 2.3 m^3 d^{-1}). This observation was consistent with evapotranspiration rates reported in short-rotation coppice forests, which are frequently higher than rates reported for grass-like crops, such as barley and grass ley (Persson and Lindroth, 1994).

Evapotranspiration rates at the field site were significantly higher than rates observed in natural wetlands with similar plant communities (section S3.1 of the SI). This was consistent with past research, which has also shown that evapotranspiration rates appear to increase in coppice forests when wastewater or sewage sludge are applied as a source of nutrients (Dimitriou and Aronsson, 2011), possibly due to increased biomass growth through nutrient enrichment (Morris et al., 2013). Higher evapotranspiration rates could be beneficial from an operational perspective, because they can drive greater flows of water into the subsurface by lowering the water table, which could increase treatment capacity.

Overland flow correlated strongly with applied flow (Spearman's $\rho = 0.88$) throughout the monitoring period. Swale-type cells had the greatest overland flow, averaging 14.2 m^3 d^{-1} compared to 6.8 m^3 d^{-1} for other cell types, while values were similar (p-values > 0.25) among all other cell types. The significantly higher overland flows in swale-type cells (compared to other cells) during earlier monitoring phases (p-value <0.001) were likely due primarily to topographical differences in the design of those cells. Isolating the impact of topography on overland flows (i.e., comparing cells A-C and cells D-F) yields a Spearman's ρ value of 0.57 − indicating a stronger correlation than observed for temperature (0.05), soil type (−0.02) or planting regime (0.25). Because of these high overland flows, swale-type cells had lower contaminant removal efficiencies than other cells (e.g., over the entire study period, 31% of applied nitrate was removed in swale-type cells compared to 75% in other cells). However, swale-type cells removed comparable masses of contaminants to other cells because of similar subsurface flows. Swale-type cells removed around 100 kg N $cell^{-1}$ of nitrate-N compared to an average removal of 112 kg of N $cell^{-1}$ in all other cell types. In the final monitoring phase, mass removal of nitrate-N in the subsurface was greatest in swale and riparian type cells with 191 and 242 g N $cell^{-1}$ d^{-1} removed in those cells respectively, compared to 131 and 180 g N $cell^{-1}$ d^{-1} in wet meadow cells with a fine and coarse topsoil type respectively.

3.3. *Implications for design and operation of horizontal levees*

The design of constructed wetlands involves tradeoffs among a variety of considerations, of which construction and operational costs, space requirements, and contaminant removal capabilities are typically most important (Kadlec and Wallace, 2009). Other considerations (Section 3.3.3), such as habitat quality, public benefits (Knight, 1997) and control of disease vectors (Knight et al., 2003) can also drive design decisions. For horizontal levees, subsurface flow capacity appears to have the most significant effect on contaminant mass removal. If sand and gravel needed for the subsurface is not readily available onsite, the purchase and transport of coarse material could increase construction costs, though possibly only marginally. Our research also suggests that other design considerations, such as the plant community composition, can have significant impacts on subsurface flow capacities in these systems, thereby impacting contaminant removal.

3.3.1. *Design and operational considerations*

In the horizontal levee, contaminant removal was largely confined to subsurface flows at the beginning of the slope, while overland flow affords negligible treatment. Because the amount of treatment achieved is likely to be a critical design objective for these systems, it is essential that they be designed to pass all of the flow through a portion of the subsurface. To achieve this in full-scale systems, appropriate selection of the materials used for subsurface flow is essential. Hydraulic conductivity of potential construction materials can be well approximated using the Carman-Kozeny equation to make *a priori* estimates (Kadlec and Wallace, 2009) or with simple laboratory tests (e.g., constant head permeameter tests). These methods tend to be quite accurate: values approximated for construction materials *a priori* were 0.6−0.9 times observed values in the test facility, while values derived from falling head permeameter tests were not significantly different ($p < 0.05$) from observed values (see S3.2 of the SI).

Available fill found on a constructed wetland site is often not suitable to provide the needed subsurface flow capacity of these systems. To strike a balance between obtaining the desired subsurface flow capacities and the cost of bringing more permeable materials to the site, engineers could build a narrower treatment zone to achieve treatment within the first few meters of the horizontal levee. Beyond this initial treatment zone, overland flow is not as much of a concern because sufficient treatment will have already been achieved although ponding of surface water should be avoided because it provides potential mosquito breeding grounds. The full sloped wetlands may need to be much longer (50−100 m in length) for geotechnical reasons (e.g., to provide an appropriate level of wave attenuation) and for ecological reasons (to provide sufficient habitat area for wildlife), but the majority of the slope could be constructed using fill found onsite that is appropriate for supporting restored wetland habitat. Additional design features, such as subsurface layers constructed with coarse materials and periodic mixing trenches, could also be included in horizontal levee design to help increase subsurface flow capacities, though it is essential that designers include controls (e.g., geotextile liners) to prevent fine sediments from migrating into the pore spaces in these coarse material zones and clogging them.

In systems where it is critical that horizontal levees meet treatment objectives, continuous real-time monitoring of conductivity can be conducted at the end of the treatment zone (depending on the system configuration) with minimal additional costs or labor requirements (Zhuiykov, 2012). This would allow operators to quickly identify conditions in which overland flow occurs because water flowing over the wetland surface has a lower salinity than water in the subsurface that gets progressively concentrated by evapotranspiration. Flow equalization could precede these systems to ensure that applied flow rates can be temporarily decreased if necessary to prevent overland flow.

3.3.2. *Comparison to other types of wetlands*

Subsurface wetlands that have been built in many locations often do not provide a significant advantage over surface-flow wetlands in terms of space requirements and performance, though they are less susceptible to seasonal variability especially in temperate climates (Kadlec, 2009). Consistent with past research, seasonal climate fluctuations did not have a significant impact on contaminant removal in the horizontal levee. Contaminant removal efficiency was not correlated with ambient or water temperatures during the monitoring period (Fig. S6). However, the climate in the San Francisco Bay Area is mild and stable: the average daily temperature was 16.4 °C at the field site, with 95% of temperatures falling between 11 and 22 °C throughout the monitoring period. The relative insensitivity of contaminant removal to seasonal variations in temperature and plant growth may also be partly explained by the asynchronous seasonality of different removal mechanisms. For example, peak activity of plant uptake of nitrate and microbial removal of nitrate occur at different times of year (Kadlec and Wallace, 2009). The mechanisms of contaminant transformation in the subsurface will be investigated further in subsequent publications.

Horizontal levees can offer significant advantages over other types of constructed wetlands used for treating wastewater effluent. Horizontal levees appear to be significantly more efficient in terms of space requirements, provided that water can be directed to the subsurface. To compare area requirements across wetland types, we calculated the wetland area needed for 90% removal (A^1_{90}) of nitrate, in hectares per ($m^3\ d^{-1}$), introduced by Jasper et al. (2014b). Open-water and vegetated wetlands have seasonal A^1_{90} values ranging from around 1.2×10^{-3} ha $(m^3\ d)^{-1}$ and 3.4×10^{-3} ha $(m^3\ d)^{-1}$ respectively in the summer to greater than 6×10^{-3} ha $(m^3\ d)^{-1}$ in the winter (Jasper et al., 2014b). For comparison, A^1_{90} values for horizontal levees were seasonally invariable and ranged from 0.1 to 0.7×10^{-3} ha $(m^3\ d)^{-1}$ (Fig. S7). The median yearly A^1_{90} value of 0.2×10^{-3} ha $(m^3\ d)^{-1}$ for horizontal levees is significantly lower than even the most efficient summer values for open-water wetlands.

3.3.3. Additional benefits

Horizontal levees can also provide additional benefits that could make them more attractive than other types of wetlands. For example, coastal wetlands can provide storm surge protection (Shepard et al., 2011), elevation gains to keep pace with sea level rise (Morris et al., 2013), plant and animal biodiversity enhancements, restored habitat, and recreational opportunities (Ghermandi et al., 2010) and increase the resiliency of tidal marshes to sea-level rise (Beagle et al., 2019). Interviews with decision makers in the San Francisco Bay Area, where there is already awareness of the technology, revealed that horizontal levees are viewed more favorably than other nutrient control options because they provide multiple potential benefits, like sustainability and climate change resiliency (Harris-Lovett et al., 2018, 2019). Other potential benefits of horizontal levees, such as their ability to provide recreational opportunities, are discussed further in Section S7 of the SI.

Growth of native plants was rapid in the test system. Dense vegetation established on the horizontal levee within three years of construction. Native plants rapidly established and outcompeted non-natives (<2% of the surface coverage consisted of non-native plants). In riparian cells, Arroyo willows (*S. lasiolepis*) reached heights above 6 m by mid-2018 (section S1 of the SI). The rapid establishment of dense vegetation observed in this system was likely due to a combination of nutrient enrichment (Morris et al., 2013) and high plant community diversity (Grace et al., 2007), both of which can lead to greater productivity.

Consistent with past wetlands research (Knight et al., 2001), diverse fauna were also attracted to the test facility, likely due to highly varied habitat and niche complementarity created by the dense biomass growth (Grace et al., 2007). At full-scale, these benefits would likely increase because habitat diversity and quality tend to increase with wetland size (Hsu et al., 2011). At the horizontal levee, we observed diverse wildlife, including ground squirrels (*Otospermophilus beecheyi*) and other rodents, jackrabbits (*Lepus californicus*), garter snakes (*Thamnophis sirtalis*), gopher snakes (*Pituophis catenifer*) and other reptiles, amphibians, such as the Pacific tree frog (*Pseudacris regilla*), and a wide array of insects, along with a large and varied community of birds (Section S9 of the SI). These observations are in line with past research suggesting that constructed wetlands provide attractive and productive habitats (Kadlec and Wallace, 2009).

4. Conclusions

Horizontal levees can achieve significant removal of a wide range of wastewater-derived contaminants, including nutrients, pharmaceuticals and F+ coliphage, while providing other benefits, such as high-quality habitat, and increased shoreline resilience to sea-level rise. At our field site, treatment efficiency was controlled by hydrological conditions, which were the strongest predictors for the removal of a wide range of contaminants removed in these systems. Certain design parameters, such as planting regimes and soil texture, also affected the total mass of contaminants that can be removed based on their influence over maximum subsurface flow rates. Additional research is needed to develop an understanding of the mechanisms behind contaminant removal in this system, as well as how horizontal levees would function under a variety of additional other design and operational conditions, and in more variable climates.

Horizontal levees may also be useful in potable water reuse scenarios, which will likely expand in the future. Currently, there are limited options for disposal of waste streams associated with wastewater reuse (e.g., reverse osmosis concentrate streams), which tend to have low volumes but high concentrations of salts, nutrients and trace organic contaminants. Horizontal levees could be used to treat these waste streams because our results suggest these systems could handle significantly higher contaminant mass loads than we have studied. This application requires testing to ensure that differences in water matrices do not adversely impact treatment capacity, such as through stress to microbial or plant communities caused by higher salinity water. If successful, horizontal levees could continue to be appropriate multi-benefit treatment options even throughout dramatic shifts in water and wastewater management.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

CRediT authorship contribution statement

Aidan R. Cecchetti: Conceptualization, Methodology, Validation, Investigation, Formal analysis, Data curation, Writing - original draft, Visualization. **Angela N. Stiegler:** Conceptualization, Methodology, Validation, Investigation, Data curation, Writing - review & editing. **Katherine E. Graham:** Methodology, Validation, Data curation, Writing - review & editing. **David L. Sedlak:** Conceptualization, Resources, Writing - review & editing, Supervision, Project administration, Funding acquisition.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wroa.2020.100052.

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