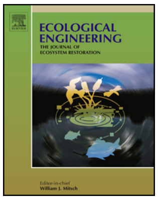
Ecological Engineering 86 (2016) 262–268

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng



Short communication

A vertical flow constructed wetland for the treatment of winery process water and domestic sewage in Ontario, Canada: Six years of performance data

Eric R. Rozema[a], Lloyd R. Rozema[b], Youbin Zheng[a],*

[a] *School of Environmental Sciences, University of Guelph, 50 Stone Road East, Guelph, ON N1G 2W1, Canada*
[b] *Aqua Treatment Technologies, 4250 Fly Road, Campden, ON L0R 1G0, Canada*

ARTICLE INFO

Article history:
Received 5 February 2015
Received in revised form 9 October 2015
Accepted 10 November 2015

Keywords:
Cold climate
Constructed wetlands
Wastewater
Winery

ABSTRACT

Wine production is a growing industry in the Canadian province of Ontario. Due to the high organic loads and large amount of suspended solids found in winery process water it is important for wineries to adopt sustainable water management practices. Constructed wetlands (CW) have been shown to be a viable option in the warmer climates of the Mediterranean and the western USA but little research has examined their feasibility in the cold climate of Ontario. The purpose of this paper was to assess six years of performance data from a vertical flow CW treating winery process water and domestic sewage at a winery in the Niagara region of Ontario with an average hydraulic loading rate (HLR) of 22.3 mm d^{-1} and an average chemical oxygen demand (COD) surface loading rate (SLR) of 34.0 g m^2 d^{-1}. The CW has four cells, each 101 m^2, and was designed to treat 16,620 L day^{-1} of wastewater consisting of winery process water and domestic sewage. The performance data were separated by season to determine the effect of temperature on treatment performance during the colder months. There was little variation in seasonal performance and the average treatment efficiencies over the six-year period were: 99% for COD, 99% for carbonaceous COD, 98% for total suspended solids, 83% for total phosphorous, 94% for total Kjeldahl nitrogen, and 85% ammonium. Effluent nitrate, total coliforms, and *Escherichia coli* concentrations were also monitored and consistently met the regulatory standards for discharge to a subsurface leaching bed. The CW system proved to be an option for treating winery wastewater.

1. Introduction

Wine production in Ontario, Canada is an expanding industry with an estimated 6600 ha dedicated to grape production in 2013 (OMAFRA, 2014). There are over 100 wineries in operation in southern Ontario (Dawson et al., 2011) and each one produces high strength wastewater. Winery process water has low nutrient content but is high in soluble organic matter with chemical oxygen demands (COD) ranging from 500 to 45,000 mg L^{-1} (Shepherd et al., 2001; Serrano et al., 2011; de la Varga et al., 2013; Masi et al., 2015). The organic load is made up of sugars, alcohols, phenols, acids, tannins and ligands all of which have variable degradation rates (Masi et al., 2015). Winery process water can also have low pH and contain large amounts of suspended solids (TSS) with concentrations reaching up to 7300 mg L^{-1} (Serrano et al., 2011; de la Varga et al., 2013). Many wineries in Ontario are open to the public for wine tastings and other events and as a result the wastewater produced by wineries also contains domestic sewage. Conventional options are ineffective for the treatment of winery wastewater because of the large organic load, as well as the fluctuating quantity and quality of the wastewater, which depends on the season and production schedule of the winery (Strong and Burgess, 2008; Serrano et al., 2011).

Constructed wetlands (CW) are engineered treatment systems designed to facilitate the biological, chemical, and physical processes that occur in natural wetlands; and they have been proven to be a viable option for the treatment of winery wastewater in certain regions (Strong and Burgess, 2008; Serrano et al., 2011; de la Varga et al., 2013). The majority of the literature reports on CWs at wineries in areas with temperate or arid climates, for example, California (Shepherd et al., 2001; Grismer et al., 2001, 2003), Italy (Masi et al., 2002), Spain (Mena et al., 2009; Serrano et al., 2011; de la Varga et al., 2013) and South Africa (Mulidzi, 2007, 2010). However, the use of CWs at wineries in cooler, continental climates

---

* Corresponding author.
E-mail address: yzheng@uoguelph.ca (Y. Zheng).

http://dx.doi.org/10.1016/j.ecoleng.2015.11.006
0925-8574/

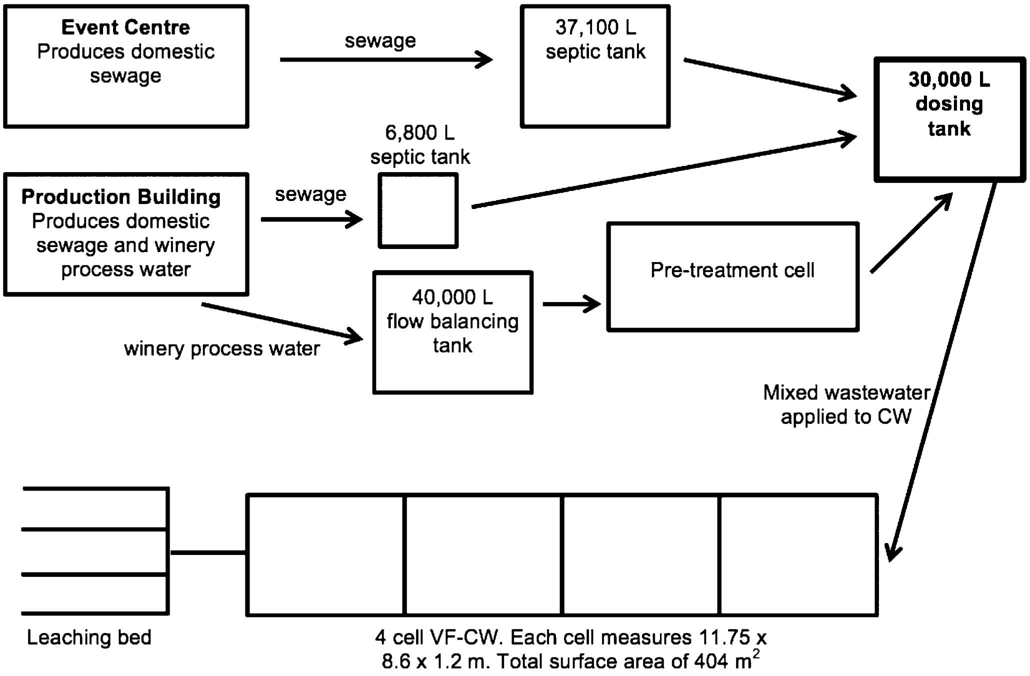


Fig. 1. Schematic diagram of the winery operations and the vertical flow constructed wetland treatment system.

in active viticulture regions such as southern Ontario, Canada, has received little attention, even though there are ~20 to 30 in use in this area. Therefore, the objectives of this study were to assess the year-round performance of a vertical flow (VF) CW at a winery in southern Ontario treating winery process water and domestic sewage.

Six years of performance data were collected from June 2008 to December 2013. The data were separated by growing season (GS; the six warmest months of the year) and non-growing season (NGS; the six coolest months) to assess the effect of temperature on treatment performance. Influent and effluent water samples were analyzed for COD, carbonaceous biochemical oxygen demand (CBOD), TSS, total phosphorous (TP), total Kjeldahl nitrogen (TKN), ammonium, nitrate, total coliforms, and *Escherichia coli*.

2. Materials and methods

2.1. Site description

The VF CW was located at a winery in Niagara-on-the-Lake, ON, Canada, an area with over 25 wineries that serve as a major tourist attraction for the region. The average annual temperature is 9 °C and the average temperatures for the GS and NGS are 17.1 °C and 1.4 °C, respectively. The average monthly precipitation is 39.6 mm and is 55.5 mm and 23.8 mm for the GS and NGS, respectively (Environment Canada, 2014). The climate is classified as humid continental (Dfb) according to the Köppen–Geiger classification system (Peel et al., 2007).

The CW was designed and installed by Aqua Treatment Technologies (Ontario, Canada) in 2008. The CW was designed to treat a maximum of 4250 L day^{-1} of winery process water and 12,435 L day^{-1} of domestic sewage for a total flow of 16,620 L day^{-1}. The CW system has four identically sized cells, 11.75 m × 8.6 m × 1.2 m and a separate 18 m × 8 m × 1.2 m pre-treatment cell (Fig. 1). The total surface area of the CW (not including pre-treatment cell) was 404 m^2 and the volume is 485 m^3. The cells were lined with a PVC liner and filled with a 5–10 mm gravel and sand mix. The top 30 cm of the cells were covered with a peat moss and sand mix and planted with *Typha latifolia* L. (broadleaf cattail) and *Schoenoplectus tabernaemontani* (C.C. Gmel.) Palla (softstem bulrush). To facilitate denitrification cell 3 contained wood chips and the water level was higher than the other cells to reduce the presence of oxygen because denitrifying bacteria require an anoxic environment as well as a carbon source. The water levels of cells 1, 2, and 4 were kept at ~0.4 m and cell 3 was kept at ~0.8 m.

The wastewater was pumped to the CW from a 30,000 L concrete storage tank, referred to as the ‘dosing tank’. The dosing tank contains water from three different sources: winery process water from the production building, a small amount of domestic sewage from the production building (no more than 250 L day^{-1}), and domestic sewage from the winery event building housing a kitchen and a public wine tasting venue (Fig. 1). The ratio of domestic sewage to winery process water fluctuated due to the nature of wine production and it was affected by the season. For example, during the summer months more people visit the winery for tastings and special events leading to an increased amount of domestic sewage but less process water is produced. In the fall after harvest wine productions begins and the processes of crushing, pressing and washing produces large volumes of wastewater. In general there were larger volumes of less concentrated wastewater produced in the NGS.

The winery process water was stored in a 40,000 L concrete flow-balancing tank before entering a pre-treatment wetland cell. A timer controlled pump in the tank limited the flow into the pre-treatment cell to a maximum of 4250 L day^{-1}. The pre-treatment cell has a similar design and plumbing system as the CW cells, it was filled with gravel and can be easily cleaned out if it becomes clogged. It acted to reduce the organic load and TSS of the wastewater before it flowed via gravity to a pumping station that pumped to the dosing tank. The sewage from the production building entered a 6800 L septic tank and a gravity overflow sent the water to the same pump station as the winery process water, from there it was eventually pumped to the dosing tank. The sewage from the event building was stored in a 37,100 L septic tank and entered the dosing tank via a gravity overflow (Fig. 1). The wastewater in the dosing tank was then applied to the cell 1 of the CW by a timer controlled pump which dictates the volume added to the CW.

Table 1
Minimum detectable limits (MDL; mg L^{-1} unless indicated) for the parameters tested.

Parameter	MDL
CBOD	<2
COD	<8
TSS	<2
TKN	<0.05
NH_4^+	<0.1
NO_3-N	<0.05
TP	<0.03
E. coli	<2 CFU 100 mL^{-1}
Total coliforms	<2 CFU 100 mL^{-1}

Each cell was flooded on the surface by dosing manifolds. There were two sets of dosing manifolds in each cell, one designated for warm weather and one for cold weather. The warm weather manifolds rested on the surface of the wetland while the cold weather manifolds were buried approximately 45 cm below the surface, which prevented them from freezing and allowed for year round operation.

The water flowed vertically down through the wetland media and then was directed to pump chambers by 10 cm perforated pipe at the bottom of each cell. From the pump chambers the water was pumped to the next cell or was discharged if in the last cell. The treated water was discharged to a sub-surface leaching bed consisting of nineteen 30 m trenches.

2.2. Sampling and analysis

Quarterly grab samples were taken from the influent and effluent water. The influent samples were taken from the line that doses cell 1 from the dosing tank. Effluent samples were taken at a sampling point from the pipe that discharges to the leaching bed. For each sampling event three 500 mL containers and one 250 mL were filled. The samples were immediately put in a cooler with ice and shipped to SGS Environmental Services (Lakefield, ON) by overnight courier. The samples were analyzed for COD, CBOD, TSS, TP, TKN, ammonium, nitrate, total coliforms, and *E. coli* according to Standard Methods (1995).

The timing of the sampling varied slightly but in general one sample was taken during spring, summer, fall, and winter, except for 2008, in which only three samples were taken in June, September, and December. The ‘winter’ sample for 2012 was actually taken in January of 2013 but it was included in the 2012 data set for simplicity.

When the samples were taken between May and October they were considered to be in the growing season (GS) and when they were taken between November and April they were labeled non-growing season (NGS). This allowed for a comparison of CW performance during the six warmest months (GS) and the six coldest months (NGS) for this region.

Influent and effluent concentrations of all parameters are presented and the removal efficiencies of COD, CBOD, TSS, TP, TKN, NH_4^+, NO_3-N are discussed as percent reductions and log reductions for *E. coli* and total coliforms. The average influent and effluent concentrations and removal efficiencies for each year and season were calculated. When effluent results are below minimum detectable limits (MDL) the percent concentration reductions are listed as >99% to indicate that near complete removal has occurred. Table 1 provides a list of the MDLs for each parameter. *t*-tests were conducted using SAS (v. 9.0, SAS Institute Inc., 1999; Cary, NC, USA) to compare the mean GS treatment efficiencies to the NGS efficiencies based on the overall reductions for each parameter to identify seasonal differences in treatment.

Flow data were estimated by tracking the amount of time the float controlled effluent pump operated for each day. The flow rate of the pump was multiplied by the time of operation to give the total volume. The inflow and outflow volumes were assumed equal.

3. Results and discussion

3.1. Treatment performance – oxygen demand

The average influent COD concentrations for the GS and NGS over the six-year period were 3043 and 2117 mg L^{-1}, respectively, and the average effluent concentrations were 6 and 14.8 mg L^{-1} (Table 2). The average treatment efficiencies of the CW were high, regardless of season, at 98.9% for both GS and NGS (Table 2). Lower temperatures do not often affect the breakdown of organic matter in CWs (Kadlec and Reddy, 2001; Werker et al., 2002) and this was also the case for this system. The removal of CBOD was also highly efficient, with average treatment efficiencies over 99% for both the GS and NGS (Table 2). The Ontario Ministry of Environment and Climate Change (MOECC; OWRA, 1990) set a discharge limit of 10 mg L^{-1} for CBOD and the CW was consistently able to meet the requirement.

COD loading often determines the design and sizing of a CW when treating winery wastewater because of the high concentrations found in winery wastewater. Shepherd et al. (2001) recommend keeping the influent COD concentrations below 5000 mg L^{-1} to avoid compromising treatment efficiency. This was achieved for this system with the use of the pre-treatment cell and by storing and mixing the winery wastewater in a dosing tank containing domestic sewage from the septic tanks (Fig. 1). However, in a few instances the influent COD concentration surpassed 5000 mg L^{-1} but there was no significant difference at $P = 0.05$ between the treatment efficiencies for that time period and periods when COD was below 5000 mg L^{-1}. The average influent COD for the 2013 GS was 5280 mg L^{-1}, with a maximum of 9500 mg L^{-1} measured in October, but the treatment efficiency was still 99%. However, there are issues with this assessment, as the retention time was not taken into account but it does suggest that this CW can handle COD concentrations higher than 5000 mg L^{-1}. High organic loads can also have phytotoxicity effects on wetlands plants (Arienzo et al., 2009; Masi et al., 2015); however, none were observed in this CW over the six-year time period.

In California Shepherd et al. (2001) examined the treatment performance of pilot sized HF-CWs and found similar COD removal rates, over 97%. Also in California, Grismer et al. (2003) assessed two different full-scale CWs treating winery wastewater year round. One system was able to remove 49–79% of the COD (SLR = 21 and 72 g m^2 d^{-1}), depending on the loading and activities occurring at the winery (i.e. non-crush vs. crush season). However, the second system was able to remove almost all COD which the authors suggest was due to lower loading rates and longer retentions times (Grismer et al., 2003).

In Italy, Masi et al. (2002) evaluated three different CW systems at three wineries. One of the systems had two 90 m^2 VF cells followed by 86 m^2 HF cell, a 148 m surface flow cell and a pond. This system was able to remove 92% of COD. Researchers in Spain evaluated a hybrid system containing hydrolytic upflow sludge bed for pre-treatment and a 50 m^2 VF cell that feed one of three parallel 100 m^2 HF cells (Serrano et al., 2011). In their study, the winery wastewater was also mixed with domestic sewage from various facilities on the premises. The CW system (pre-treatment to VF cell to one of three HF cells) achieved treatment efficiencies of 73% for COD with the VF cell reporting higher treatment efficiencies than the HF cells.

Table 2

Six years of performance data from a vertical flow constructed wetland treating winery process water and domestic sewage at a winery in southern Ontario, Canada. Average influent volumes are also presented in L day^{-1}. Data are separated by growing season (GS; the six warmest months of the year) and non-growing season (NGS; the six coolest months). Mean influent (in) and effluent (out) water quality data (mg L^{-1}) and mean treatment efficiencies (% reduced) are presented.

Parameter	2008		2009		2010		2011		2012		2013		Average	
	GS ($n=2$)	NGS ($n=1$)	GS ($n=2$)	NGS ($n=2$)	GS ($n=2$)	NGS ($n=2$)	GS ($n=2$)	NGS ($n=2$)	GS ($n=2$)	NGS ($n=2$)	GS ($n=2$)	NGS ($n=2$)	GS	NGS
CBOD														
In	756	297	1390	512	2568	891	3151	2990	322	2628	1735	1051	1653	1395
Out	0.5	MDL[a]	MDL	MDL	MDL	3	MDL	MDL	MDL	1	MDL	MDL	0.1	0.7
% red.	99.8	>99	>99	>99	>99	99.8	>99	>99	>99	99.5	>99	>99	>99	99.9
COD														
In	1165	800	2300	615	3130	1030	3780	3950	2600	4450	5280	1855	3043	2117
Out	15.5	9	6.5	9.5	4	16.5	6	9.5	MDL	26	4	18	6	14.8
% red.	94.5	98.9	99.7	98.2	99.7	98.2	99.9	99.7	>99	99.4	99.6	98.8	98.9	98.9
TSS														
In	214	212	418	38	229	106	207	290	130	236	795	183	332	178
Out	2	5	MDL	MDL	2.5	2.5	1.5	1	4	6	6	3	2.7	2.9
% red.	98.1	97.6	>99	>99	99.1	97.4	98.9	99.8	95.1	93.8	96.6	97.4	98.0	97.7
TKN														
In	17.6	1	35.4	3.5	17.6	4.7	22.3	47.6	410.8	10.1	49.5	16.7	92.2	13.9
Out	1	MDL	MDL	MDL	0.1	MDL	MDL	MDL	0.85	MDL	0.75	0.25	0.45	0.04
% red.	37.5	>99	>99	>99	99.3	>99	>99	>99	99.9	>99	95.3	92.6	88.7	98.8
NH_4^+														
In	1.05	1.1	1.95	0.2	1.3	0.2	5.4	MDL	1.0	1.4	2.4	2.55	2.18	0.91
Out	0.15	0.1	MDL	MDL	MDL	MDL	MDL	MDL	0.65	MDL	0.25	MDL	0.18	0.02
% red.	72.1	90.9	>99	>99	>99	>99	>99	–	19.4	>99	72.2	>99	72.7	98.2
NO_3-N														
In	MDL	MDL	MDL	MDL	MDL	0.65	0.06	0.07	MDL	0.23	MDL	MDL	0.01	0.16
Out	6.2	0.54	0.18	1.21	1.82	1.14	0.8	0.79	1.03	0.21	2.16	1.13	2.03	0.83
% red.	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Influent volume														
L day^{-1}	4809	7205	3965	6881	4962	8999	9886	14,719	10,969	15,499	7242	13,120	6972	11,070

[a] Below minimum detectable limit.

Table 3
Total phosphorous concentrations of the influent (in) and effluent (out) water samples and the treatment efficiencies (% reduction) of a vertical flow constructed wetland treating winery process water and domestic sewage at a winery in southern Ontario, Canada.

Date	In (mg L^{-1})	Out (mg L^{-1})	% Reduction
14-Jun-08	0.32	MDL[a]	>99
09-Sep-08	4.93	MDL	>99
29-Dec-08	0.35	MDL	>99
20-Mar-09	0.06	MDL	>99
23-Jun-09	3.52	MDL	>99
21-Sep-09	2.04	MDL	>99
22-Dec-09	1.36	MDL	>99
06-Apr-10	0.7	MDL	>99
05-Jul-10	7.82	0.04	99.5
30-Sep-10	4.88	MDL	>99
29-Dec-10	2.8	0.2	92.9
29-Mar-11	1.58	0.07	95.6
20-Jun-11	3.06	MDL	>99
26-Oct-11	6.77	0.38	94.4
13-Dec-11	8.16	0.42	94.9
03-Mar-12	8.67	0.43	95.0
10-Jul-12	2.33	0.31	86.7
19-Sep-12	4.04	0.17	95.8
30-Jan-13	0.82	0.88	−7.3
07-Mar-13	0.2	0.43	−115.0
17-Jul-13	3.79	0.92	75.7
22-Oct-13	16.5	0.19	98.8
10-Dec-13	7.7	0.31	96.0

[a] Below minimum detectable limit.

3.2. Total suspended solids

The average influent TSS concentration was 332 and 178 mg L^{-1} for the GS and NGS, respectively; and the effluent concentrations were 2.7 and 2.9 mg L^{-1} (Table 2). The effluent TSS concentrations were consistently lower than the 10 mg L^{-1} limit set by the MOECC. The treatment efficiencies for both time periods were over 97% with no observed difference in performance in the colder months. This is expected because the physical mechanisms (filtration and settling) associated with TSS removal are not affected by temperature (Werker et al., 2002).

As reported in the literature CWs are capable of removing TSS from winery wastewater (Shepherd et al., 2001; Masi et al., 2002, 2015; Grismer et al., 2003; Serrano et al., 2011; de la Varga et al., 2013). However, some type of pre-treatment is recommended to prevent clogging and issues with overloading CWs with solids (Shepherd et al., 2001; Grismer et al., 2003; Serrano et al., 2011). There were no observed issues with clogging in this CW as the pre-treatment cell helped to manage the influent TSS concentrations.

3.3. Nitrogen and phosphorous

TKN, ammonium, nitrite, and nitrate concentrations were monitored for the influent and effluent water and the total N (TN; sum of TKN, ammonium, nitrite, and nitrate) removal was assessed. A portion of the N came from the domestic sewage component of the wastewater in various forms. The MOECC set TN discharge limits at 21.25 mg L^{-1} and nitrate limits at 17 mg L^{-1} and the CW consistently met the requirements. For almost every sampling event the nitrite concentrations were below detectable levels and therefore the data are not presented or discussed in detail.

Average influent concentrations of TKN for the GS and NGS were 92.2 and 13.9 mg L^{-1}, respectively, and effluent concentrations for the GS and NGS were 0.45 and 0.04 mg L^{-1} (Table 2). During the GS the average TKN removal efficiency was 88% and during the NGS it was 99% (Table 2), with no difference at $P = 0.05$. Influent ammonium concentrations were relatively low at 2.18 and 0.91 mg L^{-1} for the GS and NGS, respectively, and average effluent concentration for the GS was 0.18 mg L^{-1} and 0.02 mg L^{-1} for the NGS. The removal efficiencies were 73 and 98% for the GS and NGS, respectively (Table 2), but these are not different at $P = 0.05$.

Similarly, Shepherd et al. (2001) and Serrano et al. (2011) report significant N concentration reductions in CWs, ~70% and ~50%, respectively. Serrano et al. (2011) attribute the majority of the N removal to the VF cell in their system but observed a strong correlation between treatment efficiency and influent concentrations. They found that when N influent concentrations were low the effluent concentrations would often increase, but in general effluent concentrations were still relatively low, on average they were below 25 mg L^{-1} TKN and 20 mg L^{-1} ammonium (Serrano et al.,

Table 4
Total coliforms and *E. coli* concentrations in the influent (in) and effluent (out) water samples (CFU 100 mL^{-1}) and the treatment efficiencies (log reduc.) of a vertical flow constructed wetland treating winery process water and domestic sewage at a winery in southern Ontario, Canada.

Date	Total coliforms			*E. coli*		
	In	Out	Log reduc.	In	Out	Log reduc.
14-Jun-08	27,000	12	3.35	2000	2	3.00
09-Sep-08	38,000	80	2.68	660	2	2.52
29-Dec-08	2	30	−1.18	2	2	0.00
20-Mar-09	500	1120	−0.35	MDL[a]	MDL	–
23-Jun-09	72,000	MDL	–	1000	MDL	–
21-Sep-09	64,000	6	4.03	660	MDL	–
22-Dec-09	560,000	40	4.15	8	MDL	–
06-Apr-10	5800	MDL	–	1400	MDL	–
05-Jul-10	20,800	2000	1.02	5600	88	1.80
30-Sep-10	4900	62	1.90	120	2	1.78
29-Dec-10	1,400,000	22	4.80	1000	MDL	–
29-Mar-11	33,000	MDL	–	20	MDL	–
20-Jun-11	73,000	MDL	–	44,000	MDL	–
26-Oct-11	20	126	−0.80	20	36	−0.26
13-Dec-11	30,000	30	3.00	1000	MDL	–
03-Mar-12	2600	MDL	–	20	MDL	–
10-Jul-12	5	110	−1.34	40	110	−0.44
19-Sep-12	48,000	380	2.10	840	2	2.62
30-Jan-13	20	20	0.00	MDL	MDL	–
07-Mar-13	1400	2	2.85	2	MDL	–
17-Jul-13	49,000	700	1.85	36,000	18	3.30
22-Oct-13	4000	640	0.80	1000	32	1.49
10-Dec-13	33,000	22	3.18	1000	MDL	–

[a] Below minimum detectable limit.

2011). Masi et al. (2002) compared three different CW systems and also report higher N removal efficiencies when the CW included a VF component.

Influent nitrate levels were very low, for the GS they averaged to 0.01 mg L^{-1} and for the NGS 0.16 mg L^{-1}. However, the effluent concentrations were higher but still much less than MOECC discharge limit of 17 mg L^{-1}, they were 2.03 and 0.83 mg L^{-1} for the GS and NGS, respectively. The higher effluent nitrate concentrations were likely due to the nitrification of ammonium that commonly occurs in the aerobic environment of VF CWs as nitrification produces nitrate from ammonium.

Total phosphorous removal decreased with time over the six-year period. During the first two years of operation all TP concentrations were consistently below detectable levels in the effluent water; however, from 2010 to 2013 the effluent concentrations of TP were almost always above zero (Table 3). TP removal efficiencies were higher initially but because the main mechanism of TP removal in CWs is adsorption (Jamieson et al., 2002; Wood et al., 2008) the removal eventually began to decrease as adsorptions sites were filled. If TP removal efficiencies continue to decrease additional treatment steps may need to be taken or the adsorption sites could be refreshed with the removal and addition of clean gravel and sand.

3.4. *Total coliforms and* E. coli

Due to the domestic sewage component of the wastewater total coliforms and *E. coli* levels were monitored. *E. coli* concentrations are monitored to assess whether fecal contamination has occurred. Total coliforms can also identify the possibility of fecal contamination and if the water is being sufficiently disinfected. In general, removal of these organisms by the CW was high, often over 1 log, however, in some cases the concentrations increased in the effluent water. The increases concentrations are not unexpected due to the natural growth of the bacteria within the CW but they are noted here because these instances greatly affected the averages calculated over the six-year period. For example, on one occasion in July 2012 the influent concentration of total coliforms was 5 CFU 100 mL^{-1} but the effluent concentration was 110 CFU 100 mL^{-1}, resulting in a −1.34 log reduction. Although, the effluent concentration of 110 CFU 100 mL^{-1} is relatively low, the average treatment efficiency for the 2012 GS was 0.38 log which does not properly represent the performance of the CW. Therefore, the raw data for total coliforms and *E. coli* are presented separately in Table 4. The Ontario Provincial Water Quality Objectives state *E. coli* concentrations should be 100 CFU 100 mL^{-1} or less to ensure the heath and quality of aquatic ecosystems are not compromised (MOEE, 1994) and the CW meets this objective for 22 of 23 sampling events with 13 events in which the effluent *E. coli* concentration was below detectable limits (Table 4).

There was no seasonal variation for influent or effluent *E. coli* or total coliform concentrations or removal efficiencies at $P = 0.05$. The diversity of removal mechanisms for bacteria in CW makes it difficult to link treatment performance to a single variable such as temperature and results in the literature are inconsistent (Werker et al., 2002) making external comparison difficult. The data show this CW is capable of removing total coliforms and *E. coli* from the wastewater but treatment efficiencies do fluctuate. Werker et al. (2002) and Smith et al. (2005) made similar conclusions and because of the frequent fluctuations it is recommended that additional disinfection technologies be used when humans are at risk of being exposed to the effluent wastewater. However, for this application the effluent water was discharged to a leaching bed and the management of bacterial contamination by the CW was considered satisfactory by the regulatory authority.

4. Conclusions

Over the six-year period the VF CW system successfully treated the wastewater combination of winery process water and domestic sewage as the effluent water quality consistently met discharge requirements. The CW removed COD, CBOD, TSS, TKN, ammonium, nitrate, *E. coli* and total coliforms. The cold climate of southern Ontario did not affect the treatment efficiencies of the CW, which were adequate regardless of season.

TP removal decreased as the CW aged suggesting that extra steps will need to be taken to manage TP in the long-term. There were fluctuations in the removal of total coliforms and *E. coli* but because the system discharged to a sub-surface leaching bed further disinfection was not needed.

This CW presents an opportunity for further research that will help improve our understanding of CW design and treatment performance. Masi et al. (2015) suggest that adding nutrients to winery wastewater could increase treatment efficiencies because some bacterial activity is nutrient limited. The sewage component of this wastewater provided nutrients to the system, however, this study was not designed to evaluate the effect of nutrient content, but this could be addressed in the future. Additionally, an in depth analysis of the wetland hydraulics would be useful in understanding the effect of retention times and flow paths on treatment performance and an investigation into the role played by the plants would also be worthwhile. Continued monitoring of the CW's performance will be useful to answer questions associated with long-term phosphorous removal as well as the overall life span of a system treating this type of high strength wastewater.

References

Arienzo, M., Christen, E.W., Quayle, W.C., 2009. Phytotoxicity testing of winery wastewater for constructed wetland treatment. J. Hazard. Mater. 169 (1), 94–99.

Dawson, H., Holmes, M., Jacobs, H., Wade, R., 2011. Wine tourism: winery visitation in the wine appellations of Ontario. J. Vacat. Mark. 17 (3), 237–246.

de la Varga, D., Ruiz, I., Soto, M., 2013. Winery wastewater treatment in subsurface constructed wetlands with different bed depths. Water Air Soil Pollut. 224 (4), 1–13.

Environment Canada, 2014. Historical Climate Data, Available at: http://climate.weather.gc.ca/ (accessed 07.04.14).

Grismer, M.E., Carr, M.A., Shepherd, H.L., 2003. Evaluation of constructed wetland treatment performance for winery wastewater. Water Environ. Res. 75 (5), 412–421.

Grismer, M.E., Tausendschoen, M., Shepherd, H.L., 2001. Hydraulic characteristics of a subsurface flow constructed wetland for winery effluent treatment. Water Environ. Res. 73 (4), 466–477.

Jamieson, T., Stratton, G., Gordon, R., Madani, A., 2002. Phosphorus adsorption characteristics of a constructed wetland soil receiving dairy farm wastewater. Can. J. Soil Sci. 82 (1), 97–104.

Kadlec, R.H., Reddy, K., 2001. Temperature effects in treatment wetlands. Water Environ. Res. 73 (5), 543–557.

Masi, F., Conte, G., Martinuzzi, N., Pucci, B., 2002. Winery high organic wastewaters treated by constructed wetlands in Mediterranean climate. In: Proceedings of the IWA 8th International Conference on Wetland Systems for Water Pollution Control, Arusha, Tanzania, September 2002.

Masi, F., Rochereau, J., Troesch, S., Ruiz, I., Soto, M., 2015. Wineries wastewater treatment by constructed wetlands: a review. Water Sci. Technol. 78 (1), 1113–1127.

Mena, J., Gómez, R., Villaseñor, J., de Lucas, A., 2009. Influence of polyphenols on low-loaded synthetic winery wastewater constructed wetland treatment with different plant species. Can. J. Civil Eng. 36 (4), 690–700.

Ministry of Environment Energy (MOEE), 1994. Policies Guidelines Provincial Water Quality Objectives. Queen's Printer for Ontario, Toronto, ON.

Mulidzi, A.R., 2007. Winery wastewater treatment by constructed wetlands and the use of treated wastewater for cash crop production. Water Sci. Technol. 56 (2), 103–109.

Mulidzi, A.R., 2010. Winery and distillery wastewater treatment by constructed wetland with shorter retention time. Water Sci. Technol. 61 (10), 2611–2615.

Ontario Ministry of Agriculture, Food, and Rural Affairs (OMAFRA), 2014. Horticultural Statistics, Available at: http://http://www.omafra.gov.on.ca/english/stats/hort/index.html (accessed 07.04.14).

1990. Ontario Water Resources Act (OWRA; R.S.O. 1990 c. O.40), Retrieved from: http://www.e-laws.gov.on.ca/html/statutes/english/elaws_statutes_90o40_e.htm (accessed 09.10.15).

Peel, M.C., Finlayson, B.L., Mcmahon, T.A., 2007. Updated world map of the Köppen–Geiger climate classification. Hydrol. Earth Syst. Sci. Discuss. 4 (2), 439–473.

Serrano, L., de la Varga, D., Ruiz, I., Soto, M., 2011. Winery wastewater treatment in a hybrid constructed wetland. Ecol. Eng. 37 (5), 744–753.

Shepherd, H.L., Grismer, M.E., Tchobanoglous, G., 2001. Treatment of high-strength winery wastewater using a subsurface-flow constructed wetland. Water Environ. Res. 73 (4), 394–403.

Smith, E., Gordon, R.J., Madani, A., Stratton, G., 2005. Pathogen removal by agricultural constructed wetlands in cold climates. J. Environ. Inf. 6 (1), 46–50.

Standard Methods, 1995. Standard Methods for the Examination of Water and Wastewater, 19th ed. American Public Health Association/American Water Works Association/Water Environment Federation, Washington, DC.

Strong, P.J., Brugess, J.E., 2008. Treatment methods for wine-related and distillery wastewaters: a review. Bioremediation J. 12 (2), 70–87.

Werker, A., Dougherty, J., McHenry, J., Van Loon, W., 2002. Treatment variability for wetland wastewater treatment design in cold climates. Ecol. Eng. 19 (1), 1–11.

Wood, J.D., Gordon, R., Madani, A., Stratton, G.W., 2008. A long term assessment of phosphorus treatment by a constructed wetland receiving dairy wastewater. Wetlands 28 (3), 715–723.