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Breakthrough of cyanobacteria in bank filtration

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ABSTRACT

The removal of cyanobacteria cells in well water following bank filtration was investigated from a source water consisting of two artificial lakes (A and B). Phycocyanin probes used to monitor cyanobacteria in the source and in filtered well water showed an increase of fluorescence values demonstrating a progressive seasonal growth of cyanobacteria in the source water that were correlated with cyanobacterial biovolumes from taxonomic counts ($r = 0.59$, $p < 0.00001$). A strong correlation was observed between the cyanobacterial concentrations in the lake water and in the well water as measured by the phycocyanin probe ($p < 0.001$, $0.73 \leq r^2 \leq 0.94$). Log removals from bank filtration estimated from taxonomic counts ranged from $0.96 \pm (0.5)$ and varied according to the species of cyanobacteria. Of cyanobacteria that passed through bank filtration, smaller cells were significantly more frequent in well water samples ($p < 0.05$) than larger cells. Travel times from the lakes to the wells were estimated as 2 days for Lake B and 10 days for Lake A. Cyanobacterial species in the wells were most closely related to species found in Lake B. Thus, a travel time of less than 1 week permitted the breakthrough of cyanobacteria to wells. Winter samples demonstrated that cyanobacteria accumulate within bank filters, leading to continued passage of cells beyond the bloom season. Although no concentrations of total microcystin-LR were above detection limits in filtered well water, there is concern that cyanobacterial cells that reach the wells have the potential to contain intracellular toxins.

1. Introduction

Cyanobacteria are prokaryote photosynthetic microorganisms that are of concern because of their ability to produce toxins and taste and odour compounds as well as disrupt drinking water treatment (Scott and Marcarelli, 2012; Zamyadi et al., 2013). The increasing proliferation of cyanobacteria is linked to the eutrophication of water bodies, notably from nitrogen and phosphorus concentrations (Taranu et al., 2015; Whitton and Potts, 2012). Cyanotoxins are most likely to occur following the accumulation of high densities of cyanobacteria in the form of blooms (Rastogi et al., 2014), with the upregulation of toxin production genes occurring when cells reach high numbers (Wood et al., 2011). Several species of cyanobacteria produce a wide range of toxic compounds with many reviews available on cyanotoxin occurrences worldwide (e.g. Gkelis and Zaoutsos, 2014; Rastogi et al., 2014). Given the increasing frequency of cyanobacteria blooms in fresh waters, there is a need for risk management strategies for drinking water suppliers to meet drinking water guidelines (e.g. Chorus, 2005) for the protection of public health (Ibelings et al., 2014; Otten and Paerl, 2015).

In regions with surficial geology appropriate for bank filtration, this technique can be an effective means of improving water quality and controlling a variety of contaminants through natural physical, chemical, and biological processes that occur during ground passage (Tufenkji et al., 2002). Table S1 (Supplementary Information) provides an overview of selected studies on the removal of algae, microbial indicators and cyanotoxins by bank filtration. Although cyanotoxins have been measured in bank filtered water with coccoid cells and filamentous cyanobacteria cell fragments (Lahti et al., 2001), other studies have not reported any cyanobacteria cells in bank filtered water with the exception of Rachman et al. (2014) in one well system with potential direct hydraulic connections to the water source. Previous research on the effectiveness of

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bank filtration for cyanobacterial removal focused on the physiochemical parameters involved in filtration, including sorption (Romero et al., 2014) or the importance of the colmation layer in the removal of cells (Harvey et al., 2015). The removal of cyanobacteria through drinking water treatment processes has shown that some species of cyanobacteria are more likely than others to pass through conventional sand filters and could result in the release of intracellular toxins into treated drinking water (Zamyadi et al., 2012c, 2013). It is unknown whether similar patterns of removal as a function of cyanobacterial species would occur in full-scale bank filtration as no studies have consistently observed the passage of cells (Supplementary Information Table S1). Furthermore, in stratified lakes, different species of cyanobacteria can be present at different depths. It is unknown whether cyanobacteria passing through bank filtration are typically benthic or planktonic species, or whether some species are more effectively removed than others.

An important short-term strategy for drinking water supplies is related to monitoring activities in support of operational decision-making. Conventional monitoring of water samples includes laboratory methods such as taxonomic analyses with cell counts and biovolume measurements and cyanotoxin analysis, triggered by the appearance of blooms in source waters or other chemical signals (Du Preez and Van Baalen, 2006; Izydorczyk et al., 2009; Newcombe et al., 2010). A more recent approach for monitoring cyanobacteria in source waters is based on *in situ* measurement of phycocyanin-specific fluorescence that can be used with real-time operational decision making to prevent cyanobacterial breakthrough to treated drinking water in surface water sources (Srivastava et al., 2013; Zamyadi et al., 2012a, 2014). However, there is a need to determine appropriate monitoring protocols for bank filtration during cyanobacterial blooms. The specific objectives were the following: 1) to estimate the efficiency of bank filtration for removing phytoplankton and bacterial indicators, 2) to determine whether there was preferential removal of certain species of phytoplankton including cyanobacteria, and 3) to provide recommendations on the use of phycocyanin probes for monitoring the fate of cyanobacteria through bank filtration.

2. Materials and methods

2.1. Study site

The study site is a bank filtration system located in Southern Quebec (Canada) near the Lake of Two Mountains. The bank filtration system consists of eight wells that pump water through the bank from two artificial lakes, A and B (Fig. 1). The lakes are not treated to control cyanobacterial blooms. An approximately 85 m wide bank separates the two lakes within which are located the wells. In order to supply the population with drinking water, seven of the eight wells (with the 8th as a stand-by pump) produce a mean daily flow rates of 8100 m^3/day. The distance of Lake A to the well field is approximately 64 m and 26 m for Lake B. Lake B was created as a result of many years of sand quarrying, and Lake A was created immediately after Lake B and remains an active sand quarry. Both lakes have a maximum depth of greater than 10 m (Richard et al., 2010).

The aquifer reservoir supplying the wells receives lateral inflows of groundwater from Lakes A and B. The local geology of the aquifer is alluvial sand filling a palaeo valley carved in the clays of the Champlain Sea (Richard et al., 2010). The aquifer section has a maximum thickness of 25–26 m, with a thickness of 21 m near the production wells. The aquifer sand is characterized by textures of medium to fine sands, with gravel in some locations. The sand bank filter materials vary from grain sizes of 0.08–2.5 mm from samples collected near the wells. Three classes of sands are present around the wells: a) a yellow sand with a low percent of silt is present in the first 6 m layer, b) a middle layer consisting of 18 m of fine beige sands, and c) the bottom layer (<2 m) consisting of fine sand and silt. The uniformity coefficient is 1.9–2.5. The mean hydraulic conductivity is 2.7×10^{-1} cm/s (Richard et al., 2010).

Results from a detailed hydrogeological model of the system were available as the model was used in the permitting process for the municipal wells (Richard et al., 2010). The regional water balance and the effects of municipal well pumping were estimated using MODFLOW-2000 (Harbaugh et al., 2000), a groundwater model under steady state and transient conditions. Measurements of water levels were conducted at 8 production wells, 9 piezometers, both Lakes A and B to calibrate the transient model. The regional water balance demonstrated that an unnamed creek acts as a regional drain entering and leaving Lake A. Lake A receives inflow from the alluvium as well as runoff from the unnamed creek that drains a predominantly agricultural watershed. No stream flows enter Lake B, although a housing development along its shoreline has limited the availability of continuous buffer strips that could mitigate lawn fertilizer loads to the lake. The vertical infiltration recharge rate to the aquifer is 310 mm/year. The horizontal velocities calculated by the model were 0.35 m/d from the edge of Lake A and 7.8 m/d from Lake B. The travel time from Lake B was 2 days and 10 days for Lake A. Results from the transient model estimate that approximately 80% of the water supply comes from the southwest (Lake A) and 20% from the northeast (Lake B) (Richard et al., 2010).

2.2. Water and sediment sampling and analysis

Lake and well water monitoring consisted of measurements conducted with: 1) an *in situ* YSI multi-parameter probe model YSI 6600 V2-4 (YSI, Yellow Spring, Ohio, USA) and 2) grab samples for phytoplankton taxonomic counts, cyanotoxin and nutrient concentrations. The *in situ* multi-probe measured phycocyanin (PC) (Relative Fluorescence Units (RFU)), Chlorophyll *a* (Chl *a,* µg/l or RFU), temperature (°C), specific electrical conductivity (mS), turbidity (NTU), DO (mg/l), and pH. A description of the probe and its use including calibration is provided in McQuaid et al. (2011).

The phosphorus and nitrogen concentrations (total phosphorus, orthophosphate and Total Kjeldahl Nitrogen (TKN)) from two measurement points in Lake A (A2 and A4), one measurement point in Lake B (B1) and well water were performed twice during the sampling period. Orthophosphates, total phosphorus, and Total Kjeldahl Nitrogen (TKN) were analyzed as per Standard Methods (APHA et al., 2012).

Six primary measurements points were selected in the lakes, including four in Lake A: A1, A2, A3, A4 and two in Lake B: B1 and B2 (Fig. 1). Sampling points A1 and A2 were located close to the bank; A3 was located near a stream discharging into Lake A, and A4 was a discharge from Lake A. The sampling locations B1 and B2 were at points where cyanobacteria blooms were detected in 2012 (Hydrophila, 2012). Samples were collected or measured *in situ* in both Lakes A and B in August, September and October 2013. The later sampling dates included additional sampling locations to include samples of surface scums collected from both lakes during blooms on the 30th of September and the 1st, 3rd and 15th of October 2013 at 0.5 m depth below the surface or from the surface scum. At each sampling point, the probe measured the profiles at depths of <0.5 m, 1.0 m, 5 m and 10 m of the water column and in the well water following bank filtration but prior to treatment. Grab samples were collected from near the surface (0.5 m) and in the well water. Analyses of microcystin-LR and taxonomic counts of phytoplankton were performed on all selected water samples. The samples were divided into two sub-samples for 1) taxonomic